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Response of *Picoides* woodpeckers to salvage harvesting of burned, mixedwood boreal forest: exploration of pattern and process

by

David Peter Stepnisky

A thesis submitted to the Faculty of Graduate Studies and Research in partial fulfillment of the

requirements for the degree of Master of Science

in

Conservation Biology

Department of Renewable Resources

Edmonton, Alberta

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The undersigned certify that they have read, and recommend to the Faculty of Graduate Studies and Research for acceptance, a thesis entitled "Response of *Picoides* woodpeckers to salvage harvesting of burned, mixedwood boreal forest: exploration of pattern and process" submitted by David P. Stepnisky in partial fulfillment of the requirements for the degree of Master of Science in Conservation Biology.

Date: Sept. 26, 2003

Abstract

I studied the effects of salvage harvesting on *Picoides* woodpecker ecology in mixedwood boreal forest, by comparing *Picoides* response among burned and non-harvested (leave), burned and salvage harvested (salvage) and unburned and non-harvested (control) sites over multiple seasons in 1999 (1st year post-fire), 2000, 2001, and 2003. Leave sites had significantly greater numbers of *Picoides*, potential prey (bark and boring beetles), and potential nesting sites (deciduous snags) throughout the study. Within leave sites, woodpeckers were most abundant in winter months compared with the breeding season, and declined annually with time-since fire. Snag densities and numbers of bark/boring beetles were positively related to woodpecker abundance, but varied by individual species, season, and year. *Picoides* abundance, potential prey abundance, and available nest sites all varied with burn severity. I recommend that areas of recently burned older forest, of varying degrees of burn severity, be maintained on the landscape to provide habitat for post-fire species.

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If I were to thank everyone who supported me through the last 4 years of my life as a graduate student, I would have to write a book. Instead of writing that book, I simply want to give thanks to a few folks who really went above and beyond to help make this project a success. Without a question, the most important person through these years has been my supervisor, Fiona Schmiegelow, who initially hired me to coordinate the Chip Lake Project, and later gave me the opportunity to continue the work as a Masters project. In addition, Fiona has given me opportunities to grow professionally, through the coordination of her Remote Areas Project, which is truly, one of the most spectacular boreal research projects that exist today. I especially appreciate the 'coffee chats' that Fiona and I have had, not only exploring topics of research, but also how to best approach the ultimate goal of 'saving the world'.

The entire concept of a salvage logging research project grew from the forward thinking of Fiona Schmiegelow, Luigi Morgantini (Weyerhaeuser Company), and Harry Stelfox (Alberta Sustainable Resource Development). I would like to thank all three of these people for the creation of the Chip Lake Study, which has helped to lead the way in new and progressive thinking towards salvage logging policies in Alberta. I would also like to thank all of my committee members (Susan Hannon, Luigi Morgantini, and John Spence) for helping in fine-tuning my research questions and treating me like a colleague through the entire thesis-writing process.

To Curtis Stambaugh, I not only owe credit to collection of more than half of the data used in this thesis, but also a big thanks for being a great friend. In the field, Curtis provided a work ethic like no other and was always keen to crack a beer and have a chat at the end of a hard day (a most admirable quality for any field-researcher). In the office, as the only other graduate student working on the Chip Lake Project, Curtis provided mental stimulation, new ideas on how to analyze data, and incredible enthusiasm when interpreting results. During cold winter days and early summer mornings a handful of hard-working field assistants made the data collection a real pleasure. In particular, Theresa Morcos, Kevin Hannah, and Christine Rice worked long hours the first winter to scout out sites and lay out the first of the research plots. These three troopers also stuck with the project for subsequent winters. I would also like to thank the many other people who spent their time in the field, including (alphabetically): Chris Applewhaite, Aaron Ball, Craig Dockrill, Warren Fleming, Tyler Flockhart, Andrew Forest, Steve Glendinning, Jesica Graham, Amanda Joynt, Joe Litke, Marg Lomow, Pat Marklevitz, Sherri Nash, Shannon Quinn, Simon Slater, Steve Smith, Curtis Stambaugh, and Maureen Wetch. I owe an extra thanks to Andy Olsevik, who helped with our tree felling for insect sampling, and through his safety and expertise, ensured that we walked away with all our limbs intact.

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Chapter 1: Research rationale and thesis introduction

1.1 Background to local and regional issues

Fire is one of the major natural disturbance agents in the boreal forest of North America. This has been well described through various fire history studies in Canada, and more specifically, in Alberta (Van Wagner 1978, Cumming 1997, Weir et al. 2000). As a result, recently burned forests are a prominent feature on boreal landscapes.

Bird communities in post-fire forests differ significantly from those in harvested areas (Schulte and Niemi 1998, Hobson and Sheick 1999). Burned forests provide unique habitat for several species of woodpeckers (notably the three-toed and black-backed woodpeckers; *Picoides arcitcus and P. tridactylus*; Murphy and Lehnhausen 1998, Hoyt and Hannon 2002) and wood-inhabiting beetles (notably those in the families Cerambycidae, Scolytidae, and Buprestidae; McCullough et al. 1998).

In spite of the proven value of post-fire forests for wildlife, forest management directives in Alberta have focused on fire suppression and salvage harvesting of recently burned forests. Although there has been a recent effort by the government to incorporate more ecologically-based planning into fire salvage, the priority is still to ensure a sustainable timber harvest. Hence, companies are encouraged to harvest large volumes of recently burned timber (Alberta Environmental Protection 2002). After a fire burns through a forested area, government and forest industry target merchantable timber, to avoid major losses that could otherwise affect annual allowable cuts within management areas. There are only limited provisions for maintenance of burned forest within salvaged areas. These provisions include leaving a proportion of snags within salvage harvested blocks as well as unburned green patches (Alberta Environmental Protection 2002).

Salvage harvesting of burned forest is conducted within the first year or two of a fire, in order to avoid loss of timber to insect damage and/or drying and subsequent cracking of the wood (Cerezke 1999). Harvesting these post-fire forests has negative impacts on certain wildlife species (notably cavity nesting birds; Hutto

1995, Caton 1996). In Europe, where fire suppression and landscape change have essentially eliminated fire from many forested landscapes, population persistence of post-fire specialist woodpeckers, beetles, and lichens is threatened, likely due to forest management and fire suppression regimes (Angelstam and Mikusinski 1994, Nilsson et al. 1995, Mikusinski et al. 2001). Presently, in boreal forests of North America, salvage harvesting of post-fire forests and fire suppression efforts are not at levels high enough to drastically reduce the amount of post-fire forest habitat available for wildlife species over large areas. However, as the boreal forest in Alberta becomes more accessible through increased road networks, fire suppression and salvage harvesting will remove an increasing proportion of recently burned forest on the landscape. As well, widespread forest harvesting is expected to significantly reduce the amount of older forest (Schmiegelow and Hannon 1993, Schnieder et al. 2003). This also has negative implications for cavity-nesting bird species and other species that depend on dead and decaying wood.

In order to better manage post-fire forests in Alberta, forest managers require increased understanding of the consequences of salvage harvesting and the importance of post-fire forests for wildlife communities. Woodpeckers in the genus *Picoides* increase in numbers dramatically after fire (Murphy and Lehnhausen 1998, Hoyt and Hannon 2002) and studies have documented fewer numbers of woodpeckers in salvage-harvested forests, when compared with burned and non-harvested forests (Caton 1996, Hitchcox 1996). However, less in known about the potential impacts of salvage harvesting on these woodpecker species in relation to other components of the burned forest (such as beetle abundance, snag densities and burn severity), nor related these effects to implications for secondary cavity nesters. Although burned forest habitat appears to be critical for primary cavity nesting species in the winter months (Kreisel and Stein 1999), it is not known how salvage harvesting affects resident secondary cavity nesting birds.

1.2 Thesis overview

I studied *Picoides* woodpeckers and associated flora and fauna for several years post-fire. Sampling was conducted in both winter and breeding seasons for the first three years post-fire, and one additional sampling interval occurred during the 5th winter post-fire. My research was conducted in the boreal mixedwood region of central Alberta, within three treatments: burned and non-harvested (leave), burned and salvage harvested (salvage) and unburned and non-harvested (control). In Chapter 2, I analyse data for the first three years post-fire, to examine differences in *Picoides* woodpecker abundance between treatments, with a particular focus on how woodpecker abundance varied within each treatment in relation to prey (bark and boring beetles) and nest-site (deciduous snags) availability, as well as how abundance of woodpeckers varied across seasons and years. In Chapter 3, I analyse woodpecker abundance and associated prey and nesting/roosting-sties in relation to burn severity for the first winter post-fire. In Chapter 4, I analyse data from 4 winter sampling intervals (up to 5 years post-fire) to assess whether abundances of secondary cavity nesters differ between treatments in the winter months, and document abundance change with time since fire. In Chapter 5, I summarise the findings of this research and discuss them in the larger context of forest management.

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Chapter 2: Variation in woodpecker abundance in burned, salvage harvested, and unburned boreal mixedwood forest

2.1 Introduction

In order to maximize timber production, salvage harvesting of burned forests has become a routine part of forest management in North America. However, recently burned forests have unique structural attributes, such as standing fire-killed trees, that provide valuable habitat for many species of birds (Hutto 1995). Bird species composition in burned forests differs markedly from that in forests that have not burned (Apfelbaum and Haney 1981, Raphael et al. 1987, Dieni and Anderson 1999), and there is increasing concern that post-fire harvesting may have long-term consequences for populations of post-fire irruptive species such as three-toed and black-backed woodpeckers (*Picoides tridactylus and P. arcticus*; Hutto 1995, Murphy and Lehnhausen 1998, Hobson and Schieck 1999, Imbeau et al. 1999, Hoyt and Hannon 2002).

Nest-site availability and food supply have been described as primary factors limiting bird populations (Newton 1998). For woodpeckers in the genus *Picoides*, snags and bark/boring beetles represent these potentially limiting resources. Previous studies have documented greater woodpecker abundance in areas of higher snag density (Dickson et al. 1983, Raphael and White 1984). Snags and damaged trees provide important nesting opportunities for hairy and downy woodpeckers (*Picoides villosus* and *P. pubescens*; Runde and Capen 1987, Schepps et al. 1999) and important foraging and drumming substrate for three-toed woodpeckers (Imbeau and Desrochers 2002). Although tree species use varies between woodpecker species and ecoregion, deciduous species seem important for nesting (Harestad and Keisker 1989, Stambaugh 2003) and conifer species for foraging (Murphy and Lehnhausen 1998). Bark and boring beetles (Coleoptera: Cerambycidae, Buprestidae, and Scolytidae) are major prey items of *Picoides* woodpeckers (Wickman 1965, Otvos and Stark 1985). Koplin (1969) speculated that increases in woodpecker numbers in burned forests

compared to unburned forests were directly related to increased food abundance, namely bark and boring beetles.

Salvage harvesting of post-fire forests has the potential to greatly reduce woodpecker abundance through removal of snags and associated bark and boring beetles. In a study of coniferous forests in northwestern Montana, Hitchcox (1996) found cavity nesting birds such as three-toed and black-backed woodpeckers nesting in greater abundance in non-harvested burned forest than salvage harvested forest and attributed the differences to availability of suitable nesting trees. Similarly, in the same coniferous forests in Montana, Caton (1996) also found greater numbers of cavity nesters in non-harvested burned forests but speculated that prey availability was more limiting than nest site availability for populations of cavity-nesting birds. Although both studies documented higher densities of woodpeckers in burned forests compared with burned then salvage harvested forests, they failed to directly quantify how these differences in abundance were related to both snags and bark and boring beetles.

It is well documented that *Picoides* woodpeckers increase in abundance immediately after fire and decline within 2-3 years. This trend has been documented both during the breeding season (Koplin 1969, Murphy and Lehnhausen 1998) and in the non-breeding season (Kreisel and Stein 1999), though seasonal differences in abundance have never been directly compared. Raphael and White (1984) suggested that areas with higher densities of snags (such as recently burned forests) might provide critical roosting and foraging areas for cavity nesting species in the winter months, when snow and cold temperatures are more energetically demanding on birds.

The purpose of this study was to examine the effects of salvage harvesting of post-fire boreal forest in Alberta on the relative abundance of *Picoides* woodpeckers, in breeding and non-breeding seasons, based on the speculation that removal of deciduous snags and bark and boring beetles would reduce nest-sites and food-supply, and ultimately limit abundance. Specifically, I tested the following questions:

1) Does woodpecker abundance vary between burned, burned and salvage harvested forest, and unburned forest?

I predicted that woodpecker abundance would be highest in non-harvested burned forest, and that this increased abundance would also be reflected in an increased number of nests relative to the other treatments.

2) Does density of potential woodpecker nest sites (deciduous snags), and potential prey (measured through an index of bark and boring beetles) vary between burned, salvaged, and unburned forest?
   I predicted that both numbers of deciduous snags and measures of bark and boring beetles would be greatest in burned forest.
3) Do numbers of deciduous snags and measures of bark and boring beetles vary with woodpecker abundance across site types?
   I predicted that woodpecker abundance would generally be positively related to deciduous snags and measures of bark and boring beetles.
4) Does woodpecker abundance vary between seasons and years post-fire in the burned forest, relative to the unburned forest and do similar changes also occur in deciduous snag densities and measures of bark and boring beetles?
   I predicted that woodpecker abundance would decrease with time since fire in the burned forest, relative to the unburned forest and woodpecker abundance would decrease from the winter season to the breeding season. I also predicted declines in deciduous snags and/or bark and boring beetles, relative to the control, mirroring those of the woodpecker declines.

2.2 Methods

2.2.1 Study area and experimental design

This study (referred to as the Chip Lake Salvage Harvesting Study) was conducted in the lower foothills ecoregion of Alberta (Strong and Leggat 1992), within the Weyerhaeuser Edson Forest Management Area north of Cynthia, Alberta (53° 25' N 115° 11' W). In May of 1998, a fire (herein referred to as the Chip Lake Fire) burned through a 14,000 ha region of the forest. In accordance with my study design, this area was salvage harvested during the winter of 1998/99. In the context

of this paper, I refer to salvage harvesting as the clear cut harvesting of recently burned forest, resulting in maintenance of little to no vertical structure.

All study sites (pre-disturbance) consisted of older (1900 origin or earlier) deciduous-dominated mixedwood forest, based on forest inventory maps provided by Weyerhaeuser Company. The dominant tree species were trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*) and white spruce (*Picea glauca*). Each site was 10 ha in size, with dimensions of 250 X 400m. A total of 21 sites were established (Figure 2.1); 14 sites were located within the burn, and 7 sites (controls) were located outside the burn perimeter. Seven of the burned sites were salvage harvested in the winter of 1998/99 (salvage sites), the other 7 remained non-harvested (leave sites). The control sites, in unburned, non-harvested forest, were established to provide a benchmark for comparisons with the salvage and leave sites, particularly in relation to temporal change. Although site selection was constrained by harvesting guidelines and availability of similar stand-types (resulting in a non-randomized block design), attempts were made to avoid pseudoreplication and erroneous study design (Hurlbert 1984) through interspersion of sites throughout the study area, equal replication of all treatments, and by ensuring that all sites were a minimum of 250 m apart. Densities of woodpeckers in post fire forests have been estimated at 0.2/ha (Murphy and Lehnhausen 1998). Suggesting that 250 m should minimise the probability of territory overlap among adjacent sites. Within the burn perimeter, all forest surrounding the study sites and containing timber suitable for harvest (i.e. older aspen and/or white spruce dominated forests) as salvage harvested (indicted by the dotted area in Figure 2.1). The number of trees left in the salvage sites was minimal.

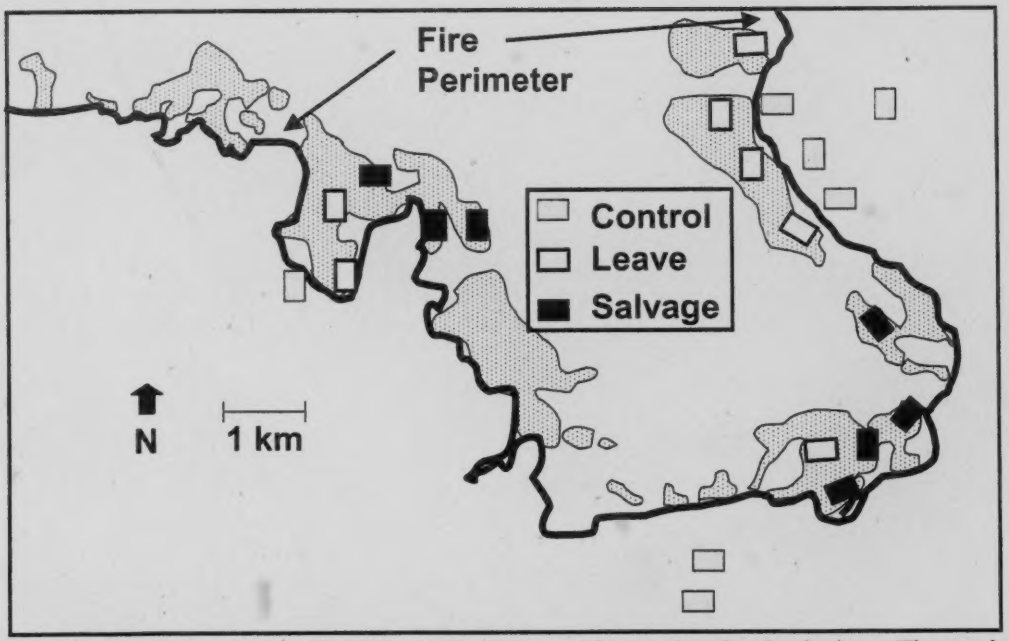

Figure 2.1. Schematic of the Chip Lake Salvage Harvesting Study design. The study area is located in the lower foothills ecoregion of Alberta, north of Cynthia, Alberta. Dotted areas indicate total area that was salvage harvested within the fire. Sites were established in the winter of 1998/99, after a fire burned through the region in the spring of 1998. Control = not burned and non-harvested, Leave = burned and non-harvested, Salvage = burned and salvage harvested.

2.2.2 Woodpecker counts

Spot mapping was used to estimate numbers of woodpeckers at each site. Amman and Baldwin (1960) tested several methods to census woodpeckers and found spot mapping (also referred to as the 'fixed plot method') to be the most effective in counting all woodpeckers in a given area. Other studies have also used this technique to determine the number of woodpeckers in a defined area (Bock and Lynch 1970, Raphael et al. 1987, Gunn and Hagan 2000).

Surveys were conducted during 3 different sampling intervals: winter, breeding season, and territory establishment, over 3 years (1999-2001). Winter

surveys were conducted from February through mid-April in 1999 (5 rounds), 2000 (8 rounds), and 2001 (4 rounds), and were concentrated around mid-day (12:00 P.M. MST). Breeding season surveys were conducted from late May through mid July in 1999 (7 rounds), 2000 (8 rounds), and 2001 (10 rounds), and occurred between 0.5 hours before sunrise and no later than 4 hours after sunrise. In addition, surveys were conducted in 2000 (5 rounds) and 2001 (4 rounds) from mid-April through mid-May during the period when woodpeckers were establishing breeding territories and occurred between 0.5 hours before sunrise and no later than 4 hours after sunrise. During the territorial establishment period and breeding seasons, 5 transect lines (50 metres apart) were walked lengthwise through each site, and observers recorded location and species of each woodpecker within the site. During the winter months, observers walked 2 transect lines along the length of each site (100m spacing between each line walked) and recorded every woodpecker seen or heard. A subset of lines was used during the winter months due to increased visibility. In all sampling intervals, observers were instructed to spend approximately 90 minutes at each site during every survey. In order to standardise effects associated with multiple observers (Verner and Milne 1990), visits to each site were rotated among observers within each sampling interval.

While efforts were made to conduct woodpecker surveys at each site for a standardised 90-minute period during each visit, specific survey lengths varied. To correct for this, all counts of woodpeckers were transformed to hourly estimates of relative abundance by calculating the mean number of woodpeckers observed in each site per hour. The results for all rounds were then averaged over the sampling interval.

Reproductive activity was assessed by searching for woodpecker nests in each of the study areas. Nests were located by listening for sounds created by excavating woodpeckers (earlier in the breeding season) and juveniles in nests (later in the season) and walking in the direction of the sound until a cavity with a woodpecker present was observed. Nest searches were conducted in all 3 years (1999-2001) in conjunction with spot-mapping surveys. Additional spot mapping surveys, with the main objective of locating nests, were conducted in all 3 years from mid-May to early

July and occurred shortly after mid-day. These additional surveys were standardised to equal search time (averaging 1.5 hours) in all sites.

2.2.3 Deciduous snag density

Vegetation plots were systematically placed at six equally spaced locations within each site (Figure 2.2). Plots were located at 40 m from the center of point count stations (established as part of another study) at 0, 120, and 240 degrees (Stambaugh 2003). Circular vegetation plots were 0.04 ha in size. In the summer of 1999, 2000, and 2001, the number of deciduous snags greater than 12 cm in diameter at breast height was recorded in each plot. This tree size class was selected for standardisation with other bird research in the region (e.g. Schmiegelow et al. 1997), and cavity nesters are not known to nest in trees below this diameter class (Hitchcox 1996). I focussed on deciduous snags because these are used by *Picoides* for nesting in our study area (Stambaugh 2003), and the value of coniferous snags for foraging (Murphy and Lehnhausen 1998) was obtained indirectly through bark and boring beetle sampling (Section 2.2.4.). Additional vegetation data were recorded within each plot, including number of trees in different DBH classes (used in conjunction with bark/boring beetle calculations; see Section 2.2.4).

Deciduous snag density at each site was estimated by summing all of the deciduous snags counted in each of the six vegetation sampling plots in each site (total area = 0.24 ha) and converting to an estimated number of deciduous snags per ha for each site in each year.

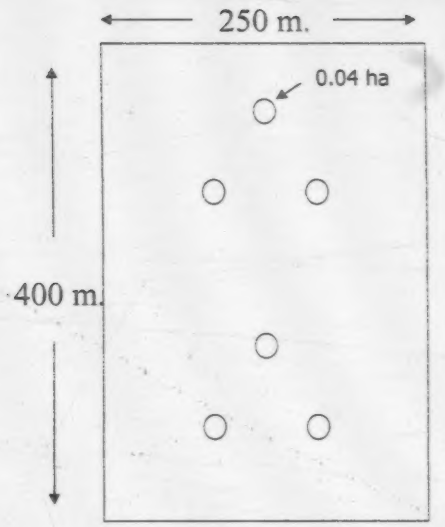

Figure 2.2. Location of vegetation plots (represented by circles) within each site within the Chip Lake Study Area. Deciduous snag densities were recorded within the vegetation plots each of three summers immediately post-fire (1999-2001), in all 21 sites.

2.2.4 Bark and boring beetle sampling

In order to quantify potential food supply for woodpeckers, numbers of bark and boring beetles were measured and an index created. In all 3 treatments in the late winter (April and May) of 1999, 2000, and 2001, intact snags (any standing dead tree that supported branches) and trees were destructively sampled, and bark and boring beetles were collected. Late winter was selected as the sampling period because larvae of bark and boring beetles should not yet have emerged from over-wintering in the wood (Ives and Wong 1988).

Each year, within all sites, three locations were randomly selected, and the nearest intact coniferous and deciduous snag or tree within a 25 m radius of each location was sampled. I chose to sample both deciduous and coniferous species, as they are known to provide habitat for different communities of bark and boring beetles (Ives and Wong 1988), all of which would be available to woodpeckers as prey. Only trees or snags within a 15-23 cm diameter at breast height size class were sampled, a procedure that was modified from protocols developed by Hoyt (2000).

This size-class restriction reduced biases that may result from sampling varying sizes of trees, which equate to different volumes of wood available for bark and boring beetles. In situations where no intact snag or tree was available within the 25 m radius, no sampling took place. For each intact snag or tree sampled, a basal section (located 0 cm from the tree base; defined as the point where the duff on the top of the forest floor ends or would have ended in the case of a burned site), a mid section (located 7 m from the tree base), and a top section (located 14 m from the tree base) was collected. Each log section collected was 45 cm in length; logs were bagged and stored in a freezer prior to lab analysis.

In the lab, both bark and boring beetles were carefully collected from each log section by first peeling off the bark to reveal surface beetles and subsequently cutting the log samples into 3.2 cm sections, and probing out any beetles found within galleries. All beetles were stored in vials containing 70% ethanol. Bark and boring beetles were counted and identified as scolytids, buprestids, or cerambycids.

An index of relative abundance of bark and boring beetles in each site (for each of scolydids, buprestids, and cerambycids separately) was calculated by summing the numbers from all beetles collected from log sections from each site in each year, dividing by the number of trees and/or snags sampled in that site, and multiplying the resulting number (a per/tree index of beetle abundance in that site) by the number of snags and trees in the 15-23 cm diameter class recorded at all vegetation plots within the same site (Section 2.2.4). This estimate (number of beetles per 0.24 ha) was then converted to a per ha value for each site (e.g. for cerambycids: Appendix 3.1 and 3.2). Estimates were separately derived for deciduous and coniferous species, but summed to obtain an average total number/ha.

2.2.5 Statistical analysis

Graphical results for woodpecker abundance are presented for untransformed data. The square root of woodpecker relative abundance was used in all analyses to satisfy assumptions of normality and equality of variance (Zar 1996). Analyses were performed on data for all sampling intervals, unless otherwise specified. Standard

errors are reported for all results in this chapter, as means of several plots or rounds were determined within each site prior to analysis by treatment. For all statistical tests, alpha was set at 0.10 to reduce the chances of committing a Type II error, as Type II errors can be more costly than Type I errors in applied conservation research (Johnson 1999). SPSS 10.0 was used to carry out statistical tests (SPSS Inc. 1999).

2.2.5.1 Relative abundance of *Picoides* woodpeckers between treatments

Due to low detection of woodpeckers in several of the sites, all species of *Picoides* (hairy woodpecker, downy woodpecker, three-toed woodpecker, and black-backed woodpecker) were combined for this analysis. Abundances of individual species of *Picoides* woodpeckers were generally consistent in how they varied between sites (Appendix 1.1) and small sample sizes made individual species analysis difficult. Other studies have combined *Picoides* woodpecker data for similar reasons (Imbeau et al. 1999).

To answer the first question of whether woodpecker abundance varied between treatments, I used a repeated measures ANOVA. Bonferroni post-hoc analysis was employed to test for differences between individual treatments. The data collected from the winter 1999 sampling interval were omitted from this analysis, because all 7 sites were sampled in the leave treatment, but only 5 sites in the control treatment and 4 sites in the salvage treatment were sampled due to ongoing salvage harvesting. Although the repeated measures ANOVA was useful in determining if time since fire was affecting woodpecker numbers, further analysis was required to answer specific questions relating to seasonal and annual variation (see Section 2.2.5.4).

Results for number of nest sites observed for all *Picoides* woodpeckers are presented, but due to small sample sizes, no statistical tests were performed.

2.2.5.2 Deciduous snag densities and bark/boring beetles between treatments

To answer my second question, Kruskal-Wallis non-parametric statistics were used to test for differences between treatments for deciduous snags, numbers of scolytids/ha, numbers of buprestids/ha, and numbers of cerambycids/ha in 1999, 2000, and 2001. Parametric statistics were not possible due to unequal variances as determined by Levene's test for homogeneity of variance.

2.2.5.3 Woodpecker abundance in relation to bark/boring beetles and deciduous snag densities

To answer my third question, I used Spearman's correlation to determine which variables were related to woodpecker abundance, then used multiple linear regression analysis to determine which of these variables (of those that showed strong correlations with woodpecker abundance) best predicted woodpecker abundance. A process of model reduction was used by removing one variable at a time (based on the variable showing the lowest significance) and performing a sums of squares F-test to determine if the reduced model was significantly different than the full model (Ramsey and Schafer 1997). If the reduced model showed no significant difference, then it was accepted over the full model, and the process repeated until only one explanatory variable remained in the model or a reduced model was rejected. Final models that best explain woodpecker abundance are presented. Each sampling interval was examined independently for relationships among variables. In addition to the *Picoides* group, individual species of woodpeckers were examined in this analysis because previous research has demonstrated that the four species of woodpeckers sampled in this study have different foraging niches (Villard and Beninger 1993, Murphy and Lehnhausen 1998). Although both beetle and deciduous snag densities were only sampled once a year, the data were used for both summer and winter analysis of woodpecker abundance, as changes between the late winter and summer were assumed to be minimal for these variables.

2.2.5.4 Annual and seasonal changes in woodpecker abundance relative to the control treatment

In order to test if woodpecker abundance varied between years and seasons in the leave and salvage treatments, relative to the control treatment, I used simple contrasts to compare changes in woodpecker abundance (Schmiegelow et al. 1997). Annual changes in relative abundance in all sites were determined for 3 different periods (1999 to 2000, 2000 to 2001 and 1999 to 2001) for 2 seasons (winter and breeding). Seasonal changes in relative abundance were determined for one seasonal period (winter to breeding) for 3 years (1999, 2000, and 2001). Although data were collected during the territorial interval, they were not included in this analysis, as it was restricted to 2000 and 2001 and was highly variable in the control treatments, making contrasts with these sites difficult to interpret. Results from comparisons between seasons were interpreted with caution due to changes in the survey intensity (which was increased in the breeding sampling interval relative to the winter sampling interval). Such variation could contribute to Type I errors. As with the first question, I combined all woodpecker species for this analysis.

Similar comparisons were made between annual changes in indices of bark and boring beetle abundance and deciduous snag densities in the leave and salvage treatments, relative to the control treatment, in order to determine how they varied from year to year.

For all analysis in this section, non-parametric statistics were preformed on all data, and for data that showed normal distributions, parametric tests were reported, as the results did not differ from non-parametric tests.

2.3 Results

2.3.1 Relative abundance of *Picoides* woodpeckers between treatments

Relative abundance of woodpeckers in the genus *Picoides* differed significantly between treatments and significantly different within treatments between sampling intervals (ANOVA, treatment: $F=33.77$, $p<0.001$, time: $F=3.57$, $p=0.012$, treatment*time: $F=2.75$, $p=0.013$, Figure 2.3). Leave sites (burned, non-harvested)

had the greatest abundance, followed by control sites (unburned, non-harvested), and the salvage sites the lowest abundance (burned, harvested) (Bonferroni: $p<0.05$ in all three post-hoc comparisons). Analyses to determine where differences existed within treatments over time (i.e. treatment*time interactions) are reported in section 2.3.4 (Annual and Seasonal Changes in Woodpecker Abundance).

Greater numbers of *Picoides* woodpecker nests were also recorded in the leave treatment compared to control and salvage harvested treatments, although sample sizes were too small for statistical analysis (Table 2.1).

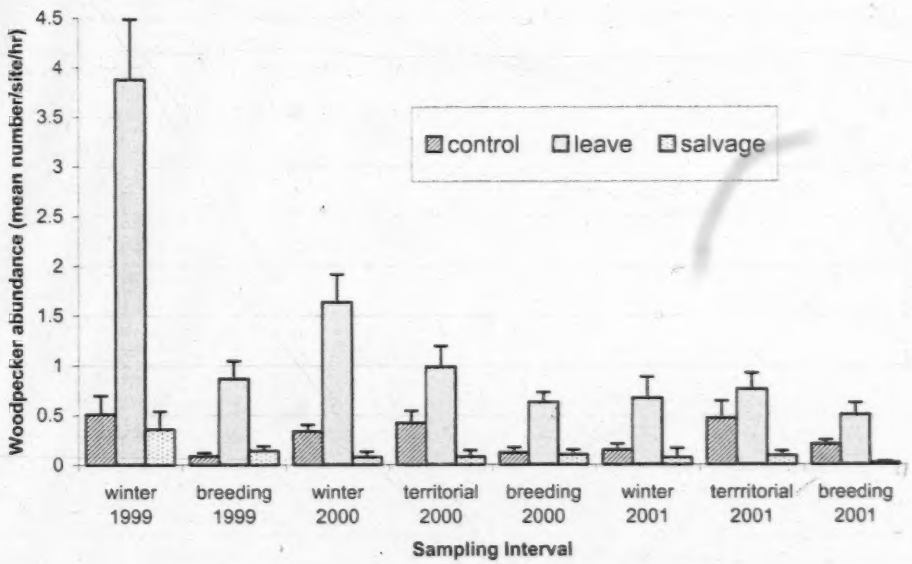

Figure 2.3. Mean relative abundance of *Picoides* woodpeckers per site (+/- SE) by treatment type for 8 sampling intervals from winter 1999 (1st year post-fire) to breeding season 2001 (3rd year post-fire) in the Chip Lake Salvage Harvesting Study Area. Control = not burned and non-harvested, Leave = burned and non-harvested, Salvage = burned and salvage harvested.

Table 2.1 Total numbers of woodpecker nests found in three different treatments within the Chip Lake Salvage Harvesting Study Area for 3 years post fire (1999-2001). Control = not burned and non-harvested, Leave = burned and non-harvested, Salvage = burned and salvage harvested.

Species	Year	Control	Leave	Salvage
Hairy Woodpecker	1999	0	2	2
	2000	1	3	0
	2001	1	2	0
Downy Woodpecker	1999	0	0	0
	2000	1	1	0
	2001	0	0	0
Three-toed Woodpecker	1999	0	2	2
	2000	0	3	1
	2001	1	0	0
Black-backed Woodpecker	1999	0	1	0
	2000	0	0	0
	2001	0	0	0
All *Picoides* Woodpeckers	1999	0	5	4
	2000	2	7	1
	2001	2	2	0

2.3.2 Deciduous snag densities and bark and boring beetles

Similar to woodpecker abundance, significant differences were detected between treatments, for all sampling intervals, for indices of buprestid abundance, indices of cerambycid abundance, and indices of scolytid abundance, and deciduous snag densities (Kruskal-Wallis, $p<0.01$ for all years for all 4 variables; Table 2.2, Figure 2.4). In all cases, the leave treatment contained the highest indices of beetle abundance and deciduous snags.

Table 2.2. Indices of bark and boring beetle abundance within trees and snags of 15-23 cm diameter class (+/- SE), by treatment, for 3 consecutive winter's post-fire (1999-2001), in the Chip Lake Salvage Harvesting Study Area. Control = not burned and non-harvested, Leave = burned and non-harvested, Salvage = burned and salvage harvested. Zero values are indicative of one or a combination of three possibilities: no trees or snags (within the 15-23 dbh class) were recorded in the vegetation plots, no trees or snags (within the 15-23 dbh class) were available to sample beetles, and/or trees or snags (within the 15-23 dbh class) were sampled but contained no beetles. $P<0.01$ for all beetles in all years (Kruskal Wallis tests for significance).

Measure	Year	Control	Leave	Salvage
Mean	1999	19.05 (12.94)	371.23 (97.75)	0.00 (0.00)
Index of	2000	0.00 (0.00)	418.25 (136.56)	10.71 (3.82)
Buprestids/ha	2001	0.00 (0.00)	52.78 (15.76)	0.00 (0.00)
Mean	1999	6.55 (4.45)	110.62 (35.63)	0.00 (0.00)
Index of	2000	5.06 (2.77)	50.99 (10.81)	0.00 (0.00)
Cerambycids/ha	2001	51.59 (28.26)	110.52 (40.55)	0.00 (0.00)
Mean	1999	1.19 (0.81)	275.89 (117.80)	0.00 (0.00)
Index of	2000	10.12 (5.54)	5454.17 (2573.46)	0.00 (0.00)
Scolytids/ha	2001	0.00 (0.00)	39.29 (12.65)	0.00 (0.00)

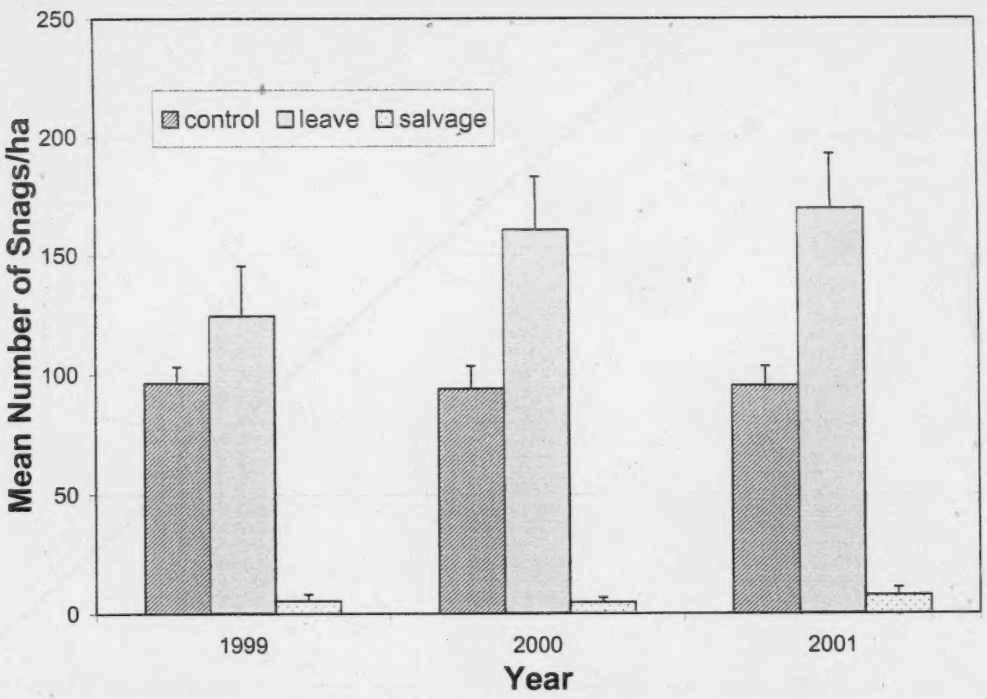

Figure 2.4. Mean number of deciduous snags/ha (+/- SE), by treatment type for 3 years post-fire (1999-2001) in the Chip Lake Salvage Harvesting Study Area. Control = not burned and non-harvested, Leave = burned and non-harvested, Salvage = burned and salvage harvested.

2.3.3 Woodpecker abundance in relation to indices of bark/boring beetle abundance and deciduous snag densities

Indices of cerambycid abundance, buprestid abundance, scolytid abundance, and deciduous snags all were significant explanatory variables for woodpecker abundance, although their relative importance differed between sampling intervals and species (Table 2.3). All relationships were positive. Black backed woodpecker abundance was not explained by deciduous snag abundance, but by all food-related variables. For all species of woodpeckers, the relative importance of any particular beetle family seemed to change with time since fire. Woodpecker abundance was explained by buprestids in the first year post fire, by cerambycids and scolytids in the second year post-fire, and by scolytids in the third year post-fire (Table 2.3.). In 1999 winter, buprestids were correlated with deciduous snags, and in 2000 (winter and breeding) scolytids were correlated with all other variables. Accordingly, results for

these sampling intervals should be interpreted with caution due to issues of co-linearity. Regression analysis was not completed for 4 sampling intervals for black-backed woodpeckers and for 3 sampling intervals for downy woodpeckers, as correlation analysis showed no relationship between any of the explanatory variables. For these 2 species, relatively low abundances likely resulted in an inability to detect relationships.

Table 2.3. Linear regression models for woodpecker species abundance for all treatments combined. Woodpecker abundance was square root transformed prior to analysis. No explanatory variables were transformed. Data are from the Chip Lake Salvage Harvesting Study Area for 3 years post fire (1999-2001).

Species and Sampling Interval[1]	Explanatory Variables[2]				P-value	R Square	Equation (with coefficients)
		Beetles					
	Decid	Bup	Cer	Sco			
Three-toed woodpecker							
1999 Winter (n=16)		+			0.01	0.38	y = 0.001(Bup) + 0.458
1999 Breeding (n=21)		+			<0.01	0.54	y = 0.001(Bup) + 0.073
2000 Winter (n=21)	+	+		+	<0.01	0.69	y = 0.002(Decid) + 4 E-6(Bup) + 5 E-5(Sco) + 0.056
2000 Breeding (n=21)			+	+	<0.01	0.46	y = 0.003(Cer) + 9 E-6(Sco) + 0.052
2001 Winter (n=21)	+				<0.01	0.33	y = 0.002(Decid) - 0.031
20001 Breeding (n=21)				+	<0.01	0.42	y = 0.003(Sco) + 0.047
Black-backed woodpecker							
1999 Winter (n=16)		+			<0.01	0.52	y = 0.001(Bup) + 0.210
1999 Breeding (n=21)[3]					-	-	-
2000 Winter (n=21)			+	+	<0.01	0.56	y = 0.004(Cer) + 2 E-5(Sco) + 0.077
2000 Breeding (n=21)[3]					-	-	-
2001 Winter (n=21)[3]					-	-	-
20001 Breeding (n=21)[3]					-	-	-
Hairy woodpecker							
1999 Winter (n=16)	+				<0.01	0.61	y = 0.005(Decid) + 0.168
1999 Breeding (n=21)		+			<0.01	0.17	y = 0.001(Bup) + 0.280
2000 Winter (n=21)	+				0.06	0.18	y = 0.001(Decid) + 0.19
2000 Breeding (n=21)	+	+	+	+	<0.01	0.80	y = 0.002(Decid) + 3 E-6(Bup) + 0.005(Cer) + 2 E-5(Sco) + 0.031
2001 Winter (n=21)	+				<0.01	0.60	y = 0.003(Decid) - 0.073
20001 Breeding (n=21)	+				<0.01	0.49	y = 0.002(Decid) + 0.068
Downy woodpecker							
1999 Winter (n=16)	+	+			<0.01	0.76	y = 0.005(Decid) + 0.001(Bup) - 0.081
1999 Breeding (n=21)[3]					-	-	-
2000 Winter (n=21)	+	+	+	+	<0.01	0.73	y = 0.002(Decid) + 3 E-6(Bup) + 0.005(Cer) + 5 E-5(Sco) - 0.046
2000 Breeding (n=21)	+				0.05	0.20	y = 0.001(Decid) - 0.019
2001 Winter (n=21)[3]					-	-	-
20001 Breeding (n=21)[3]					-	-	-
All *Picoides* woodpeckers							
1999 Winter (n=16)	+	+			<0.01	0.70	y = 0.007(Decid) + 0.001(Bup) +0.459
1999 Breeding (n=21)		+			<0.01	0.42	y = 0.001(Bup) + 0.381
2000 Winter (n=21)	+		+	+	<0.01	0.81	y = 0.004(Decid) + 0.004(Cer) + 3 E-5(Sco) + 0.195
2000 Breeding (n=21)	+		+	+	<0.01	0.64	y = 0.001(Decid) + 0.003(Cer) + 2 E-5(Sco) + 0.233
2001 Winter (n=21)	+				<0.01	0.40	y = 0.003(Decid) + 0.106
2001 Breeding (n=21)	+			+	<0.01	0.64	y = 0.002(Decid) + 0.002(Sco) + 0.140

[1] 1999 = 1st year post-fire, 2000 = 2nd year post-fire, 2001 = 3rd year post fire; n = the number of sites included in the analysis

[2] Variables used in final regression models (indicated with a '+' symbol), based on Sum of Squares F-tests between reduced and full models. All relationships were positive. Decid = Deciduous Snags, Bup = Buprestids, Cer = Cerambycids, and Sco = Scolytids

[3] No variables were correlated with woodpecker abundance for these sampling intervals; therefore no regression analysis was conducted.

2.3.4 Annual and seasonal changes in woodpecker abundance, deciduous snag density, and indices of bark/boring beetle abundance

The abundance of woodpeckers in the leave treatment, relative to the control treatment, declined significantly both between years (from the 1st to the 3rd year post fire; Table 2.4 and Figure 2.3) and between seasons (from the from winter to breeding season; Table 2.5 and Figure 2.3). The only beetle or snag variable that mirrored the significant annual declines of woodpecker abundance was buprestids (Table 2.4). Opposite to the decreases in woodpecker abundance, significant increases were observed in densities of deciduous snags (from the 1st to 3rd year post-fire).

Table 2.4. Annual changes within leave sites, relative to control sites, in mean relative abundance of *Picoides* woodpeckers, deciduous snag densities (mean number/ha/site) and indices of buprestid, cerambycid and scolytid abundance (mean number/ha/site for beetles in trees/snags of 15-23 dbh size class). Annual changes were measured for 3 intervals, 1999-2000 (1st to 2nd year post fire), 2000-2001 (2nd to 3rd year post fire) and 1999-2001 (1st to 3rd year post fire) within the Chip Lake Salvage Harvesting Study Area. Changes in abundance of woodpeckers were measured separately for winter and breeding sampling intervals. Significant results are indicated in bold type.

Treatment	**1999 to 2000**		**2000 to 2001**		**1999 to 2001**	
(Sampling Interval)	Change[1]	p-value[2]	Change[1]	p-value[2]	Change[1]	p-value[2]
Picoides Woodpeckers (Winter)	↓	**0.02***	↓	**<0.01**	↓	**<0.01***
Picoides Woodpeckers (Breeding)	↓	**0.10**	↓	0.15	↓	**0.03***
Deciduous Snags	↑	**<0.01**	↑	0.44*	↑	**0.01**
Buprestid Numbers	↑	0.54*	↓	**0.05***	↓	**0.04***
Cerambycid Numbers	↓	0.89*	↑	0.87	↓	0.63
Scolytid Numbers	↑	0.38*	↓	0.38*	↓	0.72*

[1] Change in relative abundance relative to that observed in the controls. Down arrows indicate a decline and up arrows indicate an increase.

[2] All p-values based on t-tests, unless followed by a (*), which indicates Mann-Whitney tests.

Table 2.5. Seasonal changes within leave sites, relative to control sites, in mean woodpecker relative abundance. Seasonal change in woodpecker relative abundance was measured for 3 periods, 1999 (winter to summer), 2000 (winter to summer) and 2001 (winter to summer). Significant results are indicated in bold type.

1999		2000		2001	
Change[1]	p-value[2]	Change[1]	p-value[2]	Change[1]	p-value[2]
↓	**0.01***	↓	**<0.01***	↓	0.41

[1] Change in relative abundance relative to that observed in the controls. Down arrows indicate a decline and up arrows indicate an increase.
[2] All p-values based on t-tests, unless followed by a (*), which indicates Mann-Whitney tests.

2.4 Discussion

Salvage harvesting of recently burned forests is an extensive and regular part of forest management in the northern mixedwood forests. This is the first study to look at the effects of this activity on *Picoides* woodpecker species in this forest region. I used an experimental approach to determine 1) Whether relative abundance of *Picoides* woodpeckers varied between 3 treatments (leave, salvage, and control); 2) How available prey (notably bark and boring beetles) and potential nest sites (deciduous snags) varied between the same three treatments 3) Whether these variables explained woodpecker abundance across all sites and 4) How woodpecker abundance varied across years and seasons within the leave and salvage sites (relative to the control sites). I found that *Picoides* woodpeckers, woodpecker nests, associated prey (as measured through indices of abundance of major prey species), and potential nest sites (as measured though deciduous snag densities) were all higher in the burned forests that were not salvage-harvested (leaves) when compared to the burned and salvage harvested (salvage) and unburned, non-harvested (control) forests. Both prey numbers and deciduous snag densities explained woodpecker abundance, but specific relationships varied with species, season, and year.

2.4.1 Relationship of *Picoides* woodpecker abundances to deciduous snag densities and bark/boring beetles.

Consistent with previous work conducted in burned forests (Caton 1996, Hitchcox 1996), I found significantly greater abundances of *Picoides* woodpeckers in the leave sites, followed by the control sites, followed by the salvage sites. Similar trends were observed across the treatments for potential nesting sites (deciduous snag densities) and available prey (indices of scolytid, buprestid, and cerambycid abundance). Building on previous research, I examined how these differences in potential nest sites and available prey related to abundances of different woodpecker species.

The positive relationships between deciduous snags and three-toed, hairy, and downy woodpeckers are most likely attributed to the use of snags as roosting and/or nesting sites. *Picoides* woodpeckers were found to nest in greater numbers in the leave treatment and other researchers in my study area have shown woodpeckers to preferentially nest in deciduous snags and in areas with higher mean deciduous snag densities compared with what was available in the burned sites (Stambaugh 2003). Deciduous snags as nesting trees are softer than live trees, and are thus more easily excavated by woodpeckers for nesting (Schepps et al. 1999). Black-backed woodpecker abundance was not explained by deciduous snags, suggesting that prey availability (buprestids, cerambycids, and scolytids) may be of greater importance to this species in post-fire forests.

When regression models were developed to explain woodpecker abundance, available prey (scolytids, cerambycids, and buprestids) explained abundance of three-toed woodpeckers and black backed woodpeckers, in most sampling intervals. This differed from hairy and downy woodpeckers, where available prey only explained woodpecker abundance for 2 sampling intervals. This association between three-toed/black-backed woodpeckers and bark/boring beetles is consistent with other authors, who have referred to both black-backed and three-toed woodpeckers as post-fire specialists because of their dependence on enhanced numbers of scolytids,

cerambycids, and buprestids found in burned forests (Murphy and Lehnhausen 1998, Hoyt and Hannon 2002).

Several researchers have suggested that post-fire increases in woodpecker densities are a direct result of an increase in available prey (Caton 1996, Murphy and Lehnhausen 1998). Our research supports this, but also suggests that deciduous snag density is an important additional explanatory variable for woodpecker abundance, depending on species, season, and time-since fire. Post-fire forests contain high numbers of both deciduous snags (potential nests) and beetles (potential prey), providing optimal habitat for *Picoides* woodpeckers. It is clear that salvage harvesting of burned forest greatly reduces the availability of both of these components, which translates directly into decreased woodpecker abundance.

2.4.2 Factors driving annual and seasonal changes in woodpecker abundance

Declines were recorded in woodpecker numbers from the first year to the third year post-fire in the leave treatments in both winter and breeding sampling intervals. Declines may be related to declines in bark and boring beetles (e.g. Murphy and Lehnhausen 1998, Kreisel and Stein 1999, Hoyt and Hannon 2002); however, of three beetle families sampled in my study area, only buprestids showed significant declines from the first year to the third year since fire, mirroring *Picoides* woodpecker declines. It is possible that this decline had an impact on woodpecker abundance, as buprestids inhabit portions of the wood close to the surface and have been identified as important food items for *Picoides* woodpeckers (Otvos and Stark 1985, Murphy and Lehnhausen 1998). Among variables that explained woodpecker abundance, buprestids and cerambycids were relatively more important during the first 2 years post-fire (particularly for three-toed woodpeckers and all woodpeckers combined), whereas scolytids were more important in explaining woodpecker abundance in the last 2 years of my study (2 and 3 years post-fire). It is interesting to note that there was an apparent shift from buprestids (explaining woodpecker abundance in the 1st year post fire) to cerambycids (explaining woodpecker abundance in the 2nd year post fire) to scolytids (explaining woodpecker abundance in the 2nd and 3rd year post fire).

Because all species of scolytids, buprestids, and cerambycids are highly variable in their life cycles (depending on species, temperature, and timing of entry into the wood of a tree; Ives and Wong 1988), it is difficult to relate these results directly back to the ecology of these beetles.

Relative to the control treatment, *Picoides* woodpeckers declined significantly from the winter to the breeding season in both 1999 and 2000 in the leave treatment. Otvos (1970) suggested that bark beetles become an increasingly important food source for woodpeckers in the winter months. During winter, many crawling invertebrates become inaccessible to woodpeckers as the invertebrates enter a dormant period in their life cycle. Bark and boring beetles, such as the scolytids, buprestids, and cerambycids, then become a critical food source, as the larvae are accessible to the woodpeckers immediately under the cambium of the tree and under the bark. Woodpeckers are known to alter their foraging habits in the winter months, foraging more frequently and deeper into the cambium (Conner 1979). The winter distribution of woodpeckers should therefore correspond to areas where greater accessible food resources exist. In my study area, such food resources were concentrated in the leave (burned, non-harvested) sites, and *Picoides* woodpeckers were found in significantly greater numbers in these sites for the first two winters post-fire when compared to the same sites in the breeding season. In winter months, the density of deciduous snags was also important explanatory variables for all species. Roosting sites and thermal cover may be limiting in the winter months, leading to increased selection of areas with increased snag densities. On average, models better explained woodpecker abundance (with prey numbers and deciduous snag densities) in the winter season compared to the breeding season, supporting the proposition that resources are more limiting to woodpeckers during the winter months.

2.4.3 Study limitations

Several aspects of our study design, particularly as related to the scale and extent of our sampling, may have influenced our results. First, when assessing food

as a potential limiting factor for birds, Hutto (1990) notes 1) the importance of considering how birds may perceive the food source, 2) problems of scale-of measurement, and 3) the need to measure only the standing crop. By measuring bark/boring beetles at a larger scale (through random sampling of trees and snags within 10 ha sites), I may have overlooked individual tree effects critical to the woodpeckers. For example Powell (2000) showed that, in certain circumstances, woodpeckers might respond to bark beetles at the tree level rather than at the patch level, due to elevated prey densities within those single trees. Sampling individual trees where woodpeckers were observed to forage may have resulted in stronger relationships between bark/boring beetles and woodpecker abundances. In addition to food availability, the potential influences of other limiting factors such as predation, parasites and pathogens, weather, and interspecific competition (Newton 1998) should also be investigated.

Second, depending on the intensity of burn and rot class of an individual tree or snag, numbers of bark and boring beetles infesting that tree or snag may vary (Richmond and Lejeune 1945). This may have resulted in biases unaccounted for in our data set and future research should control for variability in rot and burn severity by collecting tree and snag samples of pre-determined rot-classes or burn severities (i.e. stratified design).

Third, densities of both black backed and three-toed woodpeckers in recently burned coniferous forest have been estimated at 0.2/ha (Murphy and Lehnhausen 1998), which equates to 2 woodpeckers of each species in a 10 ha area (i.e. our sampling sites). We observed densities close to this number for three-toed woodpeckers and lower densities for black-backed woodpeckers (three-toed woodpeckers = 1.36 averaged across all leave sites in the winter of 1999, black-backed woodpecker= 0.65 averaged across all leave sites in the winter of 1999; Appendix 1.1). Thus, the scale of our study (with sampling units of 10 ha in size) may have been too small for the species of interest. Larger study units are recommended for future research. As well, we only studied woodpeckers in deciduous dominated forests. The original intent of this study was to examine the effect of salvage logging of both coniferous and deciduous dominated forests, but

maintenance of burned coniferous forest for research purposes was not approved due to concerns over timber supply. Thus, all burned conifer forest (a potentially important habitat type for post-fire specialist woodpeckers) was salvaged in our study area.

Finally, because of the adjacency of control sites to the burn perimeter, there may have been a spill over effect from leave sites into adjacent control sites. Future studies could evaluate the potential spill over effect by placing control sites at varying distances from the perimeter of the burn.

2.4.4 Relevance to forest management

In the context of forest management, my study suggests that post-fire forests (1 and 2 years post fire) are extremely valuable to *Picoides* woodpecker species, particularly in the winter months. It is clear that the availability of deciduous snags and bark/boring beetles are limiting for woodpeckers, and it is certain that removal of these elements by salvage harvesting directly relates to decreased abundance of woodpeckers in those areas. If post-fire forests are not maintained on these landscapes in the future, the long-term persistence of woodpecker populations in the boreal forests of Alberta may be negatively affected.

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Chapter 3: Importance of burn severity to woodpecker and bark/boring beetle communities in recently burned mixedwood boreal forests

3.1 Introduction

Fires in the boreal forest of North America do not burn uniformly across the landscape, but rather create a mosaic of forest types altered by both high and low intensity fires (Van Wagner 1983, Weir et al. 2000). Although a great deal of work has been undertaken to develop indices of fire intensity and burn severity (e.g., Byram 1959, Burrows 1995), ecologists rarely incorporate this information into studies that explore the impacts of fire on ecological communities (Alexander 1982). To date, the majority of avian research in post-fire forests has treated fire as a binary event (Bock and Lynch 1970, Raphael et al. 1987) assuming that bird response will be the same across the burn, regardless of variability in burn severity.

Through their apparent irruptive nature in immediate post-fire forest, several species of woodpeckers in the genus *Picoides* (most notably three-toed and black-backed woodpeckers, *Picoides tridactylus* and *P. arcticus*; Murphy and Lehnhausen 1998, Hoyt and Hannon 2002) have been labelled as post-fire specialists. In recently burned forests, *Picoides* woodpeckers primarily prey upon bark and boring beetles (Wickman 1965, Otvos and Stark 1985), and nest almost exclusively in snags with species preference varying by forest-type (deciduous snags in mixedwood forests of Alberta, Stambaugh 2003; coniferous snags in coniferous forests of Montana, Hitchcox 1996). Although the positive response of woodpeckers to increased prey and potential nest sites in recently burned forests is generally understood (Chapter 2), it is unclear how the abundance of these woodpeckers may vary across a range of burn severity within a burned landscape. In this chapter, I explore how woodpeckers in the genus *Picoides* respond in a burned landscape subject to differing burn severities, by looking not only at woodpecker abundance, but also abundance of potential prey and potential nesting/roosting sites.

Previous research on *Picoides* woodpecker abundance in relation to burn severity has been conducted at the micro-site level, examining burn severity of

individual trees used for nesting and/or foraging (Murphy and Lehnhausen 1998, Nappi et al. 2003). These studies have provided inconsistent results, in some cases showing woodpeckers selecting trees of specific burn intensities for foraging (e.g., three-toed woodpeckers selecting for lightly to moderately burned trees, black-backed woodpeckers selecting for moderately to heavily burned trees, and hairy woodpeckers selecting for heavily burned trees; Murphy and Lehnhausen 1998), and in other situations showing no clear relationships (black-backed woodpeckers; Nappi et al. 2003). Hoyt and Hannon (2002) used burn severity as one of 11 different variables to predict presence of black-backed and three-toed woodpeckers, and found three-toed woodpeckers more likely to be present in burned forests with lower densities of heavily burned trees. Vegetation characteristics other than burn severity were better predictors of black-backed woodpecker presence (Hoyt and Hannon 2002).

A limited number of studies have explored the relationship between burn severity and vegetation communities, as well as burn severity and boring beetles. Turner et al. (1997, 1999) examined vegetation communities after the 1988 Yellowstone fires, and noted that there was a great deal of variation in the vegetation responses to burn severity. When examining burn severity of individual trees, both Richmond and Lejeune (1945) and Ross (1960) found that numbers of wood-boring beetles varied significantly with burn severity. The majority of beetles were found in moderately burned trees, while very few were found in severely or lightly burned trees.

The highest densities of *Picoides* woodpeckers have been noted within the first year after a fire, and in the winter months (Murphy and Lehnhausen 1998, Chapter 2), suggesting that immediate post-fire forest may be of greatest value to these woodpecker species. For this reason, I chose to look at woodpeckers during the first winter after fire, as well as during the breeding season, one year after fire. I selected the breeding season in addition to the winter season, as woodpeckers may use burned forests differently at this time, when potential nest sites become increasingly important, and prey may not be as limiting (Conner 1979).

In this chapter, I asked the question: Are abundances of woodpecker species, bark/boring beetles, and deciduous snags related to burn severity? I hypothesised that

there should be a linear relationship between deciduous snags and burn severity and a quadratic relationship between bark and boring beetles and burn severity (Figure 3.1). This hypothesis was based on the premise that there are more snags created in a hot fire (high burn severity), and that bark and boring beetles will select sites of moderate burn severity, due to suitability of trees within those sites as hosts (Richmond and Lejeune 1945). Because of inconsistencies and lack of information from previous studies for woodpecker response to burn severity, I did not make specific predictions with regards to individual species response.

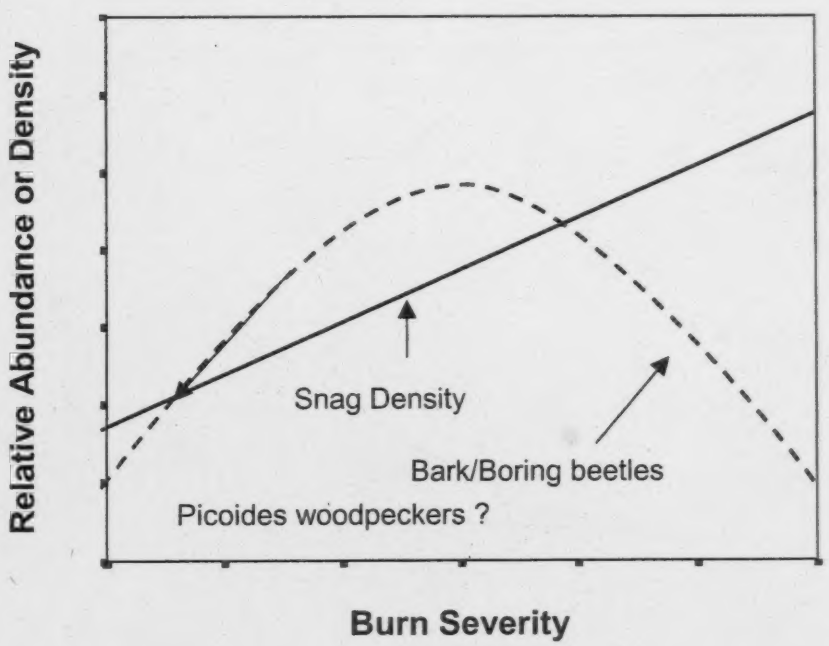

Figure 3.1. Conceptual model of the effects of burn severity on densities of deciduous snags, *Picoides* woodpecker relative abundance, and bark/boring beetles.

3.2 Methods:

3.2.1 Study area

This study (referred to as the Chip Lake Salvage Harvesting Study) took place in the lower foothills ecoregion of Alberta (Strong and Leggat 1992), within the Weyerhaeuser Edson Forest Management Area north of Cynthia, Alberta (53° 25' N 115° 11' W). In May of 1998, a fire (herein referred to as the Chip Lake Fire) burned through a 14,000 ha region of the forest. The fire varied in intensity, resulting in areas within the fire perimeter that varied from unburned (fire skips) to heavily burned (crown fire).

A total of seven sites that varied in the amount of visible fire damage were established within the fire (Figure 3.2). These sites were established as a component of a larger study, examining the effects of salvage harvesting on post-fire bird communities (see Chapter 2). All sites (pre-fire) consisted of older (1900 origin or earlier) deciduous-dominated mixedwood forest, based on forest inventory maps provided by Weyerhaeuser Company. The dominant tree species were trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*) and white spruce (*Picea glauca*). Each site was 10 ha in size, with dimensions of 250 X 400m. Areas with older spruce or aspen dominated forests were salvage harvested surrounding the 7 study sites (as indicated by the dotted area on Figure 3.2).

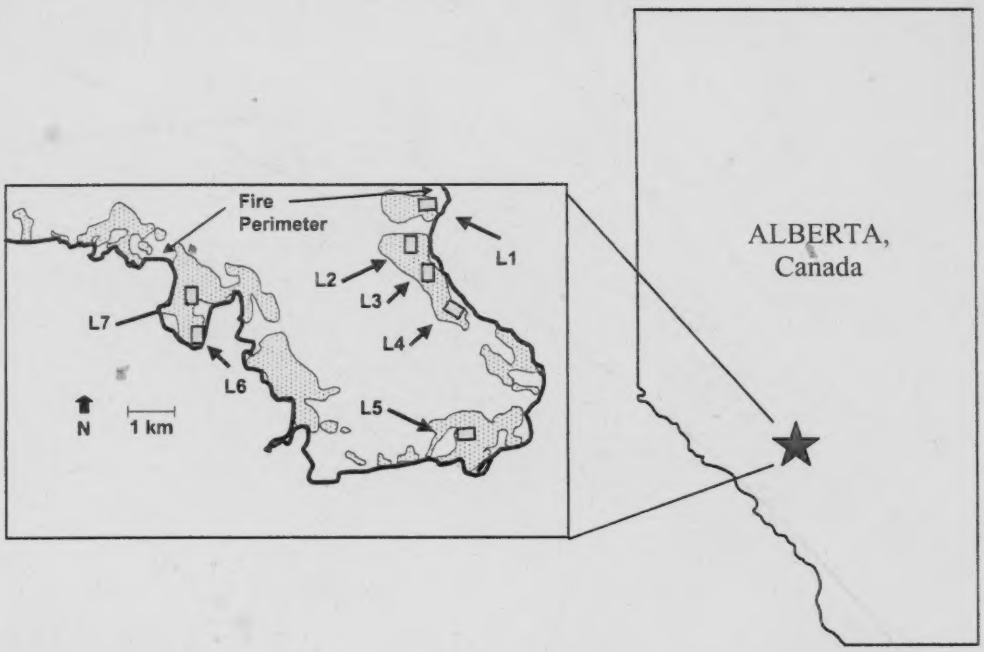

Figure 3.2. Schematic of the Chip Lake Salvage Harvesting Study design. The study area is located in the lower foothills ecoregion of Alberta, north of Cynthia. Sites were established in the winter of 1998/99, after a fire burned through the region in the spring of 1998. Each site (represented by the shaded rectangles and labelled by site number on the inset diagram) is 250 X 400 m in size. Areas surrounding the 7 burn sites were salvage logged, as indicated by the dotted area.

3.2.2 Burn severity

Vegetation plots were systematically placed at 6 locations within each site (Figure 3.3). Plots were located at 40 m from the center of point count stations (established as part of another study) at 0, 120, and 240 degrees (Stambaugh 2003). Circular vegetation plots were 0.04 ha in size. At each of these six plots, burn severity was estimated during the summer of 1999 (one year post-fire), within a 25 m radius, using a 6-point index (Table 3.1), modified from Hutto (1995), and averaged across the site.

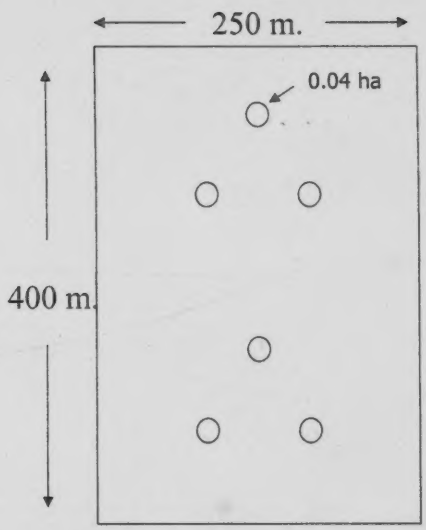

Figure 3.3. Location of vegetation plots (represented by the inner circles, each with a radius of 11.3m) within each site within the Chip Lake Study Area. Burn severity was estimated within a larger 25 m radius circle, centred in the middle of each vegetation plot.

Table 3.1. Index of burn severity used in the Chip Lake Salvage Harvesting Study area to classify sites (modified from Hutto 1995). Burn severity was measured at all plots in each site, during the summer of 1999, one year after fire.

Numerical Index of Burn Severity	Description of trees within a 25m radius
0	no burned trees
1	more than 60% of trees having green needles/leaves vs. brown or none
2	40-60% of trees having green needles/leaves vs. brown or none
3	5-40% of trees having green needles/leaves vs. brown or none
4	less than 5% of trees having green needles/leaves vs. brown or none
5	all trees having brown or no needles/leaves
6	mostly broken stumps with a few standing trees remaining

3.2.3 Deciduous snag densities

The number of deciduous snags greater than 12 cm diameter at breast height was recorded in each of the six vegetation plots within each of the seven sites (Figure 3.3). This tree size was selected for standardisation with other bird research in the region (e.g. Schmiegelow et al. 1997), and cavity nesters are not known to nest in trees below this diameter class (Hitchcox 1996). I did not examine coniferous snag

densities for two reasons: 1) deciduous snags are used by *Picoides* woodpeckers for nesting in our study area (Stambaugh 2003) and 2) the value of coniferous snags for foraging (Murphy and Lehnhausen 1998) was obtained indirectly through sampling (Section 3.2.4). Additional vegetation data were recorded within each plot, including number of trees in different DBH classes (used in conjunction with calculations; see Section 3.2.4).

Deciduous snag density at each site was estimated by summing all deciduous snags in each of the six vegetation sampling plots at each site (a total area of 0.24 ha) and converting to an estimated number of deciduous snags per ha per site.

3.2.4 Woodpecker prey

To determine indices of bark and boring beetle abundance, intact snags (any standing dead tree that supported branches) and trees were destructively sampled, and bark and wood inhabiting invertebrates were collected at each site in 1999 (one year post-fire). Late winter was selected as a sampling period because larvae of bark and boring beetles should not yet have emerged from over-wintering in the wood (Ives and Wong 1988).

Each year, within all sites, three locations were randomly selected, and the nearest intact coniferous and deciduous snag or tree within a 25 m radius of each location was sampled. I chose to sample both deciduous and coniferous species, as they are known to provide habitat for different communities of beetles (Ives and Wong 1988), all of which would be available to woodpeckers to prey upon. Only trees or snags within a 15-23 cm diameter at breast height size class were sampled, a procedure that was modified from protocols developed by Hoyt (2000). This size-class restriction reduced biases that may result from sampling varying sizes of trees, which equate to different volumes of wood available for bark/boring beetles. In situations where no intact snag or tree was available within the 25 m radius, no sampling took place. For each intact snag or tree sampled, a basal section (located 0 cm from the tree base; defined as the point where the duff on the top of the forest floor ends or would have ended in the case of a burned site), a mid section (located 7 m from the tree base), and a top section (located 14 m from the tree base) was

collected. Each log section collected was 45 cm in length; logs were bagged and stored in a freezer prior to lab analysis.

In the lab, both bark and boring beetles were collected from each log section by first peeling off the bark to reveal surface beetles and subsequently cutting the log samples into 3.2 cm sections, and probing out any bark/boring beetles found within galleries. All beetles were stored in vials containing 70% ethanol. Beetles were counted and identified down to the lowest taxa possible. Although numerous families of invertebrates were collected, only buprestids, scolytids, and cerambycids were used in analysis, as they are thought to be the most important food item for *Picoides* woodpeckers in recently burned forests (Murphy and Lehnhausen 1998).

Indices of abundance of each family of bark and boring beetle relative to each site were calculated separately by summing the total number collected from log sections from each site in each year, dividing by the number of trees and/or snags sampled in that site, and multiplying the resulting number (a per/tree estimate of numbers in that site) by the number of snags and trees in the 15-23 cm diameter class recorded at all vegetation plots within the same site (Section 3.2.3). This index (number of beetles per 0.24 ha) was then converted to a per ha value for each site. Indices were separately derived for deciduous and coniferous species but summed to obtain an average total number/ha for each site.

3.2.5 Woodpecker counts

Spot mapping was used to estimate numbers of woodpeckers at each site. Amman and Baldwin (1960) tested several methods to census woodpeckers and found spot mapping (also referred to as the 'fixed plot method') to be the most effective in counting all woodpeckers in a given area. Other studies have also used this technique to determine the number of woodpeckers in a defined area (Bock and Lynch 1970, Raphael et al. 1987, Gunn and Hagan 2000).

Surveys were conducted during 2 different sampling intervals, winter and the breeding season, in 1999 (one year after fire). Winter surveys were conducted from February through mid-April (5 rounds) and were concentrated around mid-day (12:00 P.M. MST). Breeding season surveys were conducted from late May through mid

July (7 rounds) and occurred between 0.5 hours before sunrise and no later than 4 hours after sunrise. During the breeding sampling interval 5 transect lines (50 metres apart) were walked lengthwise through each site, and observers recorded the exact location and species of each woodpecker within the site. During the winter months, observers walked 2 transect lines along the length of each site (100m spacing between each line walked) and recorded every woodpecker seen or heard. A subset of lines was used during the winter months due to increased visibility (making it possible to locate woodpeckers from further distances). In all sampling intervals, observers were instructed to spend approximately 90 minutes at each site during every survey. In order to standardise effects associated with multiple observers (Verner and Milne 1990), visits to each site were rotated among observers within each sampling interval.

While efforts were made to conduct woodpecker surveys at each site for a standardised 90-minute period during each visit, specific survey lengths varied. To correct for this, all counts of woodpeckers were transformed to hourly estimates of relative abundance by calculating the mean number of woodpeckers observed in each site per hour. The results for all rounds were then averaged over each sampling interval.

3.2.6 Statistical analysis

Because burn severity indices existed for 6 vegetation plots within each of the seven sites, I used these data to test for differences between each site. A Kruskal-Wallis non-parametric analysis of variance (ANOVA) was used, grouping burn severity indices collected at individual vegetation plots within a site. Kruskal-Wallis non-parametric tests were used due to the small numbers of samples in each data set, coupled with unequal variances.

I used correlation analysis to examine if burn severity was related to deciduous snag densities, woodpecker abundance (for hairy, downy, black-backed, and three-toed woodpeckers, and all species combined, in both winter and breeding sampling intervals), and bark/boring beetles (buprestids, scolytids, and cerambycids). When correlation analysis revealed a significant relationship, I further explored each

relationship with regression analysis. Separate regression analyses were used to examine the predicted relationships between burn severity and each dependent variable. For the comparison of deciduous snag densities and burn severity, as well as woodpecker abundance and burn severity, linear regression analysis was used. For the comparisons of bark and boring beetles and burn severity, quadratic relationships were predicted, and therefore quadratic curves were fit to the data and tested for significance, even when correlation analysis showed no strong relationship (as correlation analysis would assume a linear pattern). Linear relationships were also examined, and sums of squares F-tests were used to determine if the reduced (linear) model was significantly different than the full (quadratic) model (Ramsey and Schafer 1997). If the reduced model showed no significant difference, then it was accepted over the full model. Only the final models are presented.

For all statistical tests, alpha was set at 0.10 to reduce the chances of committing a Type II error, as Type II errors can be more costly than Type I errors in applied conservation research (Johnson 1999). SPSS 10.0 was used to carry out statistical tests (SPSS Inc. 1999).

3.3 Results

3.3.1 Burn severity

Burn severity varied both within sites and between sites within the Chip Lake Salvage Harvesting Study Area. Significant differences in burn severity were detected between the seven sites (Table 3.2; Kruskal-Wallis; $\chi^2 = 27.65$, df = 6, $p<0.001$). Two sites experienced relatively low levels of burn severity, two sites experienced high levels of burn severity, and three sites experienced moderate levels of burn severity. Sites that exhibited low burn severity typically consisted of many live trees within the forest, with light scorch marks on the bases of some of the trees. Sites with high burn severity indices were comprised of mostly dead trees, with solid black scorch marks from the base of the trees, all the way up to the crown (typical of high intensity crown fires). Moderately burned sites exhibited a great deal of variation

in burn severity within each site (i.e. between the six vegetation plots within each site).

Table 3.2. Variation in burn severity for seven burn sites in the Chip Lake Salvage Harvesting Study Area. Mean burn severity at each site represents an average of burn severities from 6 systematically placed vegetation plots per site. Sites differed significantly (Kruskal-Wallis; $\chi^2 = 27.65$, df = 6, $p<0.001$).

Burn Site #	Mean Burn severity	St. Dev.	Severity Class
4	0.2	0.2	Low
3	1.3	0.9	Low
1	2.6	1.6	Moderate
6	2.7	1.8	Moderate
5	3.8	2.0	Moderate
7	4.4	1.2	High
2	5.0	0.0	High

3.3.2 Relating burn severity to deciduous snag density, woodpecker prey and woodpeckers

There was a significant, positive linear relationship between burn severity and deciduous snag density at the seven burned sites in the Chip Lake Salvage Harvesting Study Area (Figure 3.4; $R^2 = 0.89$, $p=0.001$). If deciduous snag density is a good indicator of potential woodpecker nest sites, it follows that forests experiencing higher burn severity should have a greater abundance of potential woodpecker nest sites.

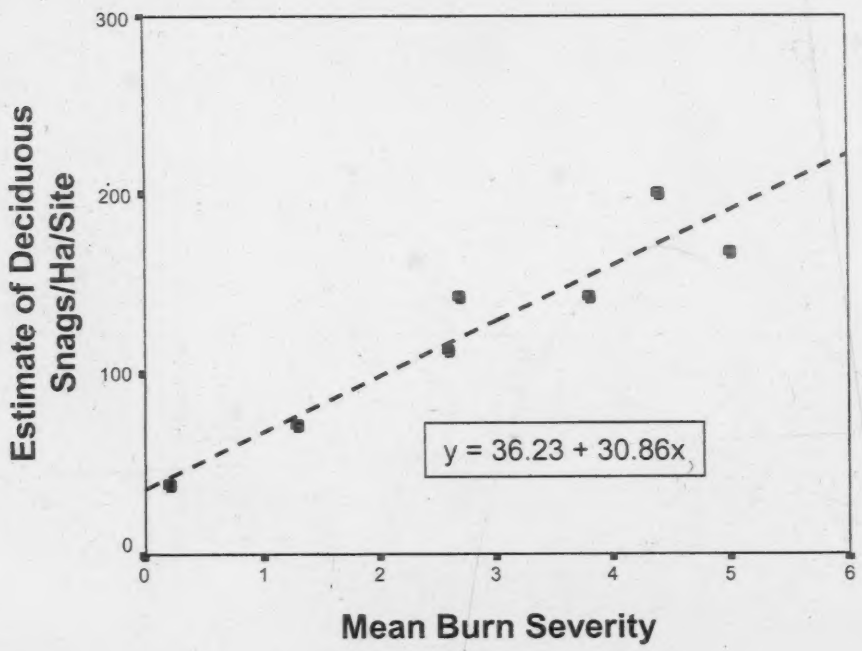

Figure 3.4. Estimate of deciduous snags (average number/ha/site) in seven different sites that varied in burn severity within the Chip Lake Salvage Harvesting Study Area. The dotted line represents the best-fit linear model (R^2=0.89, p=0.001), which indicates a significant linear relationship between snag density and burn severity. Data were collected during the summer of 1999, one year post-fire.

Indices of bark and boring beetles collected also varied a great deal across burn sites, with the majority of the beetles recorded in the moderately burned sites (Figure 3.5). Neither buprestids, scolytids, or cerambycids were present in the 2 sites with low mean burn severity. Although relatively high indices of abundance of buprestids were recorded in the 2 high burn sites, scolytids and cerambycids were either absent or found in very low numbers. While deciduous and coniferous trees were equally sampled, the majority of bark and boring beetles were collected from coniferous trees, notably white spruce (Appendix 3.3 and 3.4). Contrary to my prediction, no significant relationships (either quadratic or linear) were detected between burn severity and either scolytids or cerambycids (Spearman's correlation; scolytids r_s=0.15, p=0.75, cerambycids r_s=-0.04, p=0.93, buprestids r_s=0.67, p=0.10).

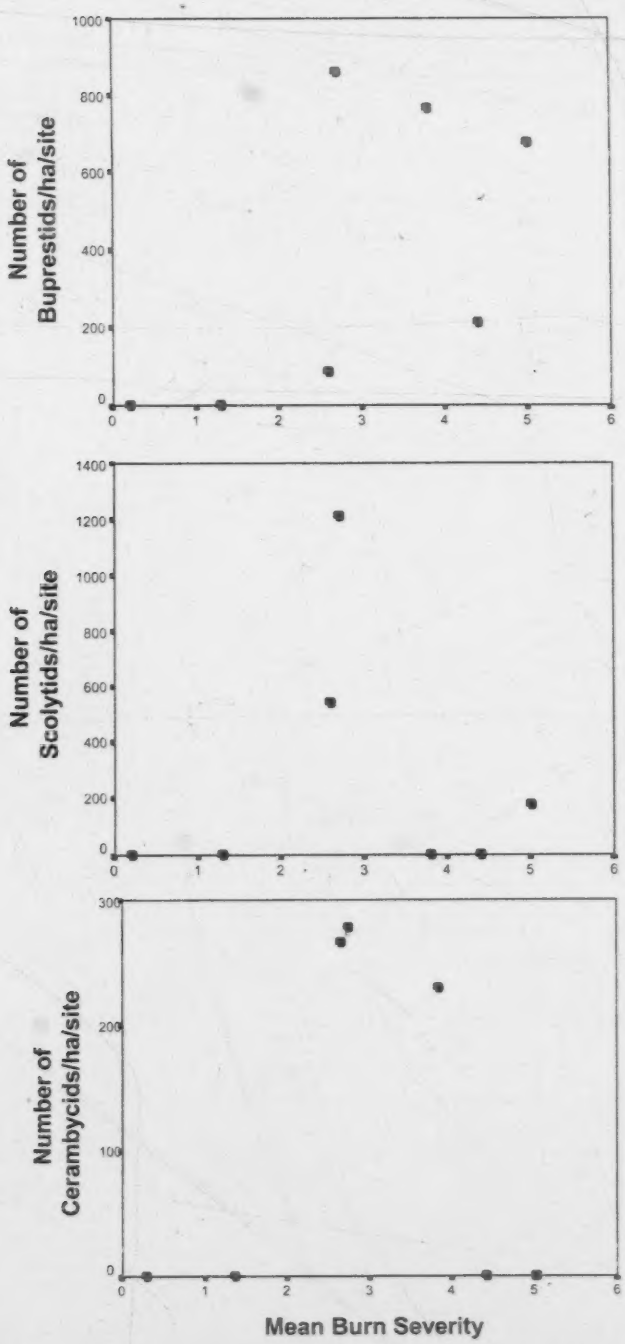

Figure 3.5. Indices of abundance of bark and boring beetles collected in seven different sites that varied in burn severity in the Chip Lake Salvage Harvesting Study Area. No significant relationships existed between burn severity and number of s. All data were collected during the late winter of 1999, one year post-fire.

Abundances of woodpeckers were highly variable, depending on species and burn severity, as well as season (Figures 3.6 and 3.7). In the winter sampling interval (Figure 3.6), a significant positive linear relationship was observed between abundance of both hairy and downy woodpeckers and burn severity (hairy woodpecker R^2=0.78, p=0.009; downy woodpecker R^2=0.81, p=0.006). Abundances of three-toed woodpeckers and black-backed woodpeckers were not related to burn severity in winter (Spearman's Correlation; three-toed r_s=0.07, p=0.88, black-backed r_s=0.46, p=0.29). During this winter sampling interval both the greatest and lowest abundances of black backed and three-toed woodpeckers were seen in moderate burn sites. For the breeding season sampling interval (Figure 3.7), the abundance of downy woodpeckers was correlated with burn severity (Spearman's Correlation; downy r_s=0.75, p=0.05), but did not show a linear pattern and linear regression analysis was not done. Downy woodpeckers were absent from 4 sites with the lowest burn severities. Neither hairy, three-toed, or black-backed woodpeckers were correlated with burn severity (Spearman's Correlation; hairy r_s=-0.21, p=0.65, three-toed r_s=0.15, p=0.75, black-backed r_s=0.41, p=0.36). It is notable that while black-backed woodpeckers were present in 6 of 7 burned sites in winter they were absent from all but one site (a high burn severity site) during the breeding season.

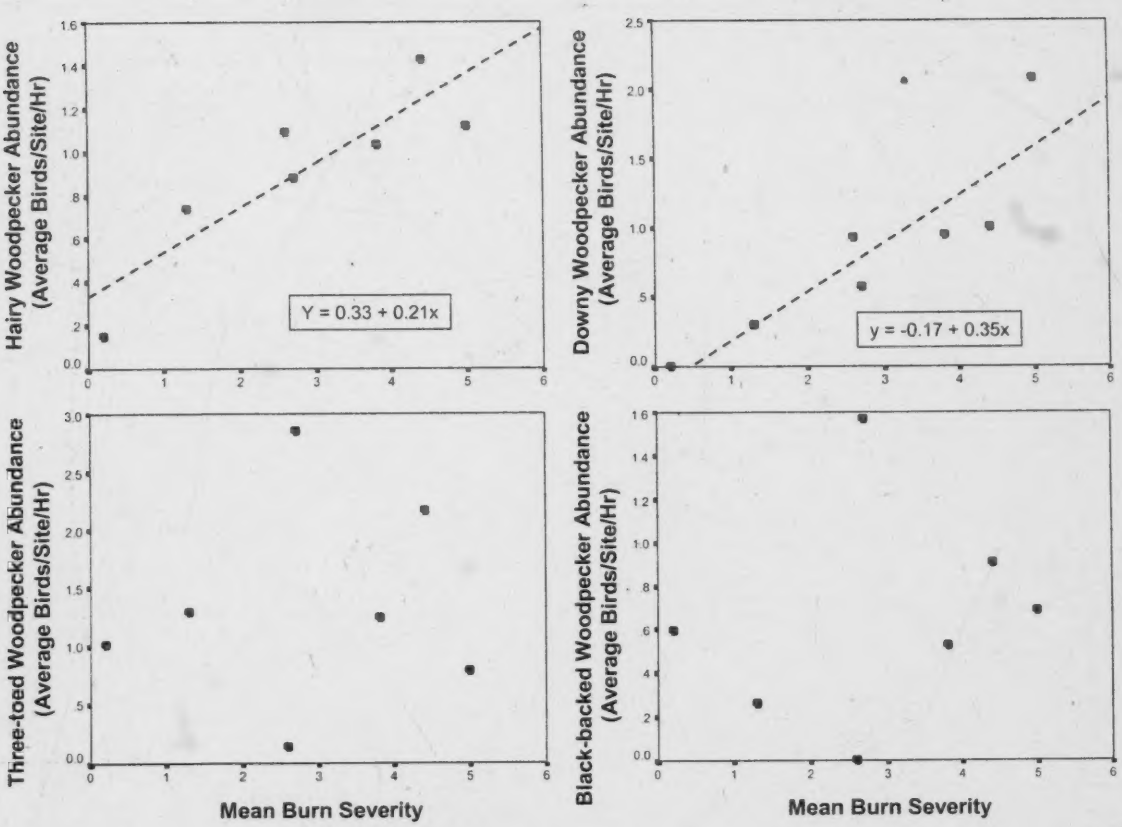
Hairy Woodpecker Abundance (Average Birds/Site/Hr)
Y = 0.33 + 0.21x
Downy Woodpecker Abundance (Average Birds/Site/Hr)
y = -0.17 + 0.35x
Three-toed Woodpecker Abundance (Average Birds/Site/Hr)
Black-backed Woodpecker Abundance (Average Birds/Site/Hr)
Mean Burn Severity
Mean Burn Severity

Figure 3.6. Relative abundance of four species of *Picoides* woodpeckers (average number/site/hr), collected during the **winter** one year post-fire, in seven different sites that varied in burn severity within in the Chip Lake Salvage Harvesting Study Area. Only hairy and downy woodpecker abundances were related to burn severity. The dotted line represents the best-fit linear model for these two species (hairy woodpecker R^2=0.78, p=0.009; downy woodpecker R^2=0.81, p=0.006).

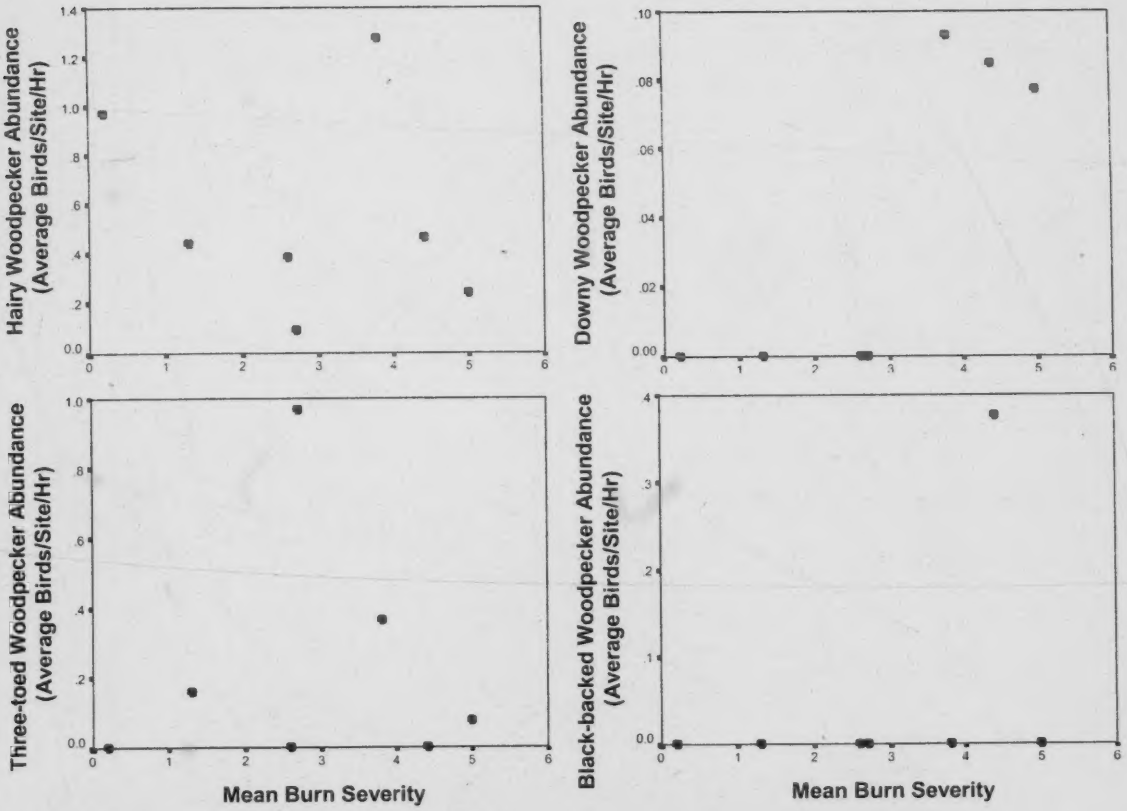
Hairy Woodpecker Abundance (Average Birds/Site/Hr)
Downy Woodpecker Abundance (Average Birds/Site/Hr)
Three-toed Woodpecker Abundance (Average Birds/Site/Hr)
Black-backed Woodpecker Abundance (Average Birds/Site/Hr)
Mean Burn Severity
Mean Burn Severity

Figure 3.7. Relative abundance of four species of *Picoides* woodpeckers (average number/site/hr), collected during the **breeding season** one year post-fire, in seven different sites that varied in burn severity within in the Chip Lake Salvage Harvesting Study Area. Burn severity did not explain woodpecker abundances.

3.4 Discussion

Previous studies of birds in burned forests have assumed fire to be a binary event, considering bird response to be uniform across the landscape. However, fires do not burn uniformly, and I suggest that birds do not respond uniformly to this disturbance. In order to gain a greater understanding of how birds vary with burn severity, I asked the question: Are abundances of *Picoides* woodpecker species, bark/boring beetles, and deciduous snags related to burn severity within a post-fire forest landscape? I studied response of: hairy, downy, three-toed, and black-backed woodpeckers in winter and breeding seasons; buprestid, cerambycid, and scolytid beetles in late winter; and deciduous snag densities in the summer in seven sites of varying burn severity within the Chip Lake Fire. Deciduous snags, hairy, and downy woodpeckers were all significantly more abundant in sites with increasing burn severity. Although not all species of woodpeckers and bark/boring beetles were statistically related to burn severity, they all showed consistently lower numbers in the sites with low burn severity relative to sites with moderate or high burn severity. Scolytid and cerambycid beetles were found to have higher indices of abundance in sites with moderate burn severity, and were absent in samples from low and high burn severity sites. Buprestid beetles were absent in the low burn sites. The results from this research are an important step in relating site-level variability in burn severity within post-fire forests to the abundance of wildlife they support.

3.4.1 Explaining responses of deciduous snag density and bark/boring beetles to burn severity

As predicted, deciduous snag densities increased significantly with burn severity. The explanation for this is obvious: as a fire burns with greater intensity (resulting in a greater severity of burned forest) it kills a greater number of trees, resulting in higher snag densities.

Based on previous literature, I had predicted that all families of bark and boring beetles would exhibit a quadratic response, with lowest numbers recorded in the low and high severity sites. Although there were no statistically significant

quadratic relationships in my study area, the majority of beetles were detected in moderately burned sites, with fewer detected in sites with high and low burn severity. Sites with lower burn severities contained mostly live trees that were presumably resistant to beetle attack, and in the high burn sites, the wood and bark of the trees may have been desiccated to the point where they were inhospitable for beetle colonization (see Richmond and Lejeune 1945).

3.4.2 Exploring responses of woodpecker abundance in relation to burn severity

The relationship of *Picoides* woodpecker abundance to burn severity varied, largely depending on species and season. Patterns were more evident in the winter season than in the breeding season, and may be a result of resources (both food and roosting sites) being more limiting during that period. This is consistent with results from Chapter 2, where abundance of woodpeckers increased in burned forests during the winter months.

During the winter, hairy and downy woodpecker abundances increased significantly with burn severity. This response mirrored that of deciduous snag densities, suggesting that nesting and/or roosting site limitations may be related to abundances of these woodpecker species. In Chapter 2, I found deciduous snags to be the only variable that explained both hairy and downy woodpecker numbers during this same winter sampling period. Neither three-toed nor black-backed woodpeckers exhibited significant relationships with burn severity during the winter season. Other studies have suggested that three-toed and black-backed woodpeckers are drawn to the burned forests because they respond to increased prey availability within the burn (Murphy and Lehnhausen 1998). Higher abundances of these 2 species of woodpecker were observed in sites with very high levels of bark and boring beetles (the sites with moderate burn severity), however, this relationship was not consistent across all sites. I suggest that in the winter months, three-toed and black-backed woodpeckers may be using the forest at scales that were not considered in this study. For example, woodpeckers may roost in sites with low burn intensity (e.g., thermal regulation gained from forests with increased live-tree cover), and fly to adjacent

patches of forest with higher burn intensities to forage. Powell (2000) concluded that although increased prey was the primary cause for increases in black-backed woodpeckers in recently burned forests, it was not the dominant factor that determined where black-backed woodpeckers were located. He suggested that woodpeckers settle within the burn based on factors other than prey. I believe that, in general, woodpeckers are attracted to burned areas, but then exhibit species specific settlement patterns: hairy and downy woodpeckers may select areas with higher densities of deciduous snags (i.e. higher burn severity) and three-toed and black-backed woodpeckers settlement patterns may be dictated more by intraspecific competition.

During the breeding season, I found no significant relationships between burn severity and any species of woodpecker. Three-toed woodpeckers were the only of the 4 species that displayed a qualitative pattern, and this could be partially explained by bark/boring beetles. Three-toed woodpeckers showed higher abundances in the moderate sites, where greater numbers of bark and boring beetles were also found. This is consistent with my results from Chapter 2, suggesting that cerambycids were explaining abundances of this species during the breeding season. Hoyt and Hannon (2002) were able to predict three-toed woodpecker presence using burn severity, and found these woodpeckers to be associated with vegetation of lower burn severity, suggesting that this was also related to higher abundances of bark beetles. It is difficult to directly compare my results with the results of Hoyt and Hannon (2002) for 2 reasons: 1) The forest type in my study (deciduous dominated mixedwood) differed from the coniferous dominated forests studied by Hoyt and Hannon (2002) and 2) Our studies used different measures for burn severity. I suggest that future studies of burn severity and wildlife management adopt a standard index for measuring burn severity, in order to make results more comparable across studies. These indices should include other quantitative forms of measuring burn severity (such as height of burn on tree boles or depth of burn in duff).

3.4.3 Study limitations

The most limiting factor in this analysis was the small number of burned sites within our study area. Although some patterns emerged, many more sites are required to determine the relationships or lack-there of between burn severity and woodpecker abundance. There was also a great deal of variability within my research sites, particularly in the moderately burned sites. Within each of the 10 ha sites, there was a mosaic of lightly to heavily burned sections. Woodpeckers may have been attracted to areas of differing burn severities within each site, which was overlooked by using a mean index of burn severity, thus considering each site to be uniformly burned. Future studies should consider mapping patches of differing burn severity within a fire to look at how species respond at different spatial scales.

Depending on the intensity of burn and rot class of an individual tree or snag, numbers of bark and boring beetles infesting that tree or snag may vary (Richmond and Lejeune 1945). This may have resulted in biases unaccounted for in our data set and future research should control for variability in rot and burn severity by collecting tree and snag samples of pre-determined rot-classes or burn severities within each site (i.e. a stratified design).

3.4.4 Relevance to forest management

It is clear that forest fires do not leave a simple legacy of burned or unburned forest. There is considerable variability in burn severity within post-fire forests, and both researchers and managers should not consider fire as a binary event when considering the resultant effects on wildlife communities. Traditionally, salvage harvesting operations in burned forests target areas with low to moderate burn severities, for the primary reason that this wood is less damaged than in high severity burns. This practice should be revisited if forest managers are interested in maintaining a range of natural variability on the landscape. By harvesting the low and moderately burned sites, important components of the post-fire forest ecosystem could be lost, particularly in relation to bark/boring beetles and black-backed/three-

toed woodpeckers (species that are found in greater abundances in moderately burned sites).

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Chapter 4: Where do resident birds reside? A look at winter abundance of secondary cavity nesting birds in an experimental post-fire and post-salvage harvest study

4.1 Introduction

Understanding the effects of landscape change on wildlife is critical for resource managers working in boreal forest regions of Canada, where rapid and widespread industrial development has the potential to drastically alter the structure and composition of forests (Schmiegelow and Hannon 1993). Although substantial research has been conducted on the effects of forest harvesting on breeding bird communities in these forests (e.g. Schmiegelow et al. 1997, Schieck et al. 2000), very little research has examined related effects on resident bird communities in the winter months (Schmiegelow et al. 1997), and no research has examined the effects of salvage harvesting (the harvest of recently burned forests) on cavity nesting species in the winter.

The winter climate within the boreal forest region of Canada is extremely harsh, and lack of roost sites and available food may greatly limit populations during this season (Newton 1998). For cavity nesting resident bird species (species that use existing tree cavities for nesting and roosting), selection of roost sites may be one of the most important behavioural adjustments for winter survival, primarily for heat conservation (e.g., the black-capped chickadee; *Poecile atricapillus*; Smith 1991). Limited information exists on roost site selection by secondary cavity nesting species; however, strong associations have been shown to exist between the presence of primary cavity nesting bird species (species that excavate their own cavities for nesting and roosting) and secondary cavity nesting birds (Martin and Eadie 1999). Raphael and White (1984) found that secondary cavity nesters readily used holes created by primary cavity nesters for nesting in the summer months, and speculated that both snags and cavities may be critical for survival of secondary cavity nesting species in the winter months.

High abundances of *Picoides* woodpeckers are found in boreal forests for 2-3 years following fire (Murphy and Lehnhausen 1998, Hoyt and Hannon 2002, Chapter 2). These primary cavity-nesting birds also have higher densities of nests in these burned forests compared to salvage harvested (burned and then harvested) and unburned forests (Hitchcox 1996). Assuming that roosting sites are of critical importance to secondary cavity nesters in the winter months, it follows that recently burned forests should support high abundances of secondary cavity nesters in the winter months, and salvage harvested forests would support much lower numbers. Further, secondary cavity nesters should continue to increase for the first few years after a fire, as a direct result of increased numbers of cavities created each year by *Picoides* woodpeckers.

Here, I asked the following two questions:

1) Are winter abundances of secondary cavity nesting nesters (SCN's) affected by salvage harvesting? I compared winter SCN abundances in (1) unburned and non-harvested sites (control), (2) burned and non-harvested sites (leave) and (3) burned and salvage harvested sites (salvage), predicting that the salvage sites would support significantly fewer SCN's compared to both the leave and control sites.
2) Do abundances of SCN's increase within burned forests for the first few winters after fire? I examined change in SCN abundance over 5 winters post-fire in the leave sites (relative to the controls) with the prediction that SCN's would increase because of annual excavation of new cavities by *Picoides* woodpeckers, resulting in annual increases in roost sites available to SCN's.

4.2 Methods

4.2.1 Study area and experimental design

This study (referred to as the Chip Lake Salvage Harvesting Study) was conducted in the lower foothills ecoregion of Alberta (Strong and Leggat 1992), within the Weyerhaeuser Edson Forest Management Area north of Cynthia, Alberta (53° 25' N 115° 11' W). In May of 1998, a fire (herein referred to as the Chip Lake Fire) burned through a 14,000 ha region of the forest. In accordance with my study design, this area was salvage harvested during the winter of 1998/99. In the context of this paper, I refer to salvage harvesting as the clear cut harvesting of recently burned forest, resulting in maintenance of little to no vertical structure.

All study sites (pre-disturbance) consisted of older (1900 origin or earlier) deciduous dominated mixedwood forest, based on forest inventory maps provided by Weyerhaeuser Company. The dominant tree species were trembling aspen (*Populus tremuloides*), balsam poplar (*Populus balsamifera*) and white spruce (*Picea glauca*). Each site was 10 ha in size, with dimensions of 250 X 400 m. A total of 21 sites were established (Figure 2.1); 14 sites were located within the burn, and 7 sites (controls) were located outside the burn perimeter. Seven of the burned sites were salvage harvested in the winter of 1998/99 (salvage sites), the other 7 remained non-harvested (leave sites). The control sites, in unburned, non-harvested forest, were established to provide a benchmark for comparisons with the salvage and leave sites, particularly in relation to temporal change. Although site selection was constrained by harvesting guidelines and availability of similar stand-types (resulting in a non-randomized block design), attempts were made to avoid pseudoreplication and erroneous study design (Hurlbert 1984) through interspersion of sites throughout the study area, equal replication of all treatments, and by ensuring that all sites were a minimum of 250 m apart. Within the burn perimeter, all forest surrounding the study sites and containing timber suitable for harvest (i.e. aspen and spruce dominated forests) was salvage harvested (indicted by the dotted area in Figure 2.1). The number of trees left in the salvage sites was minimal.

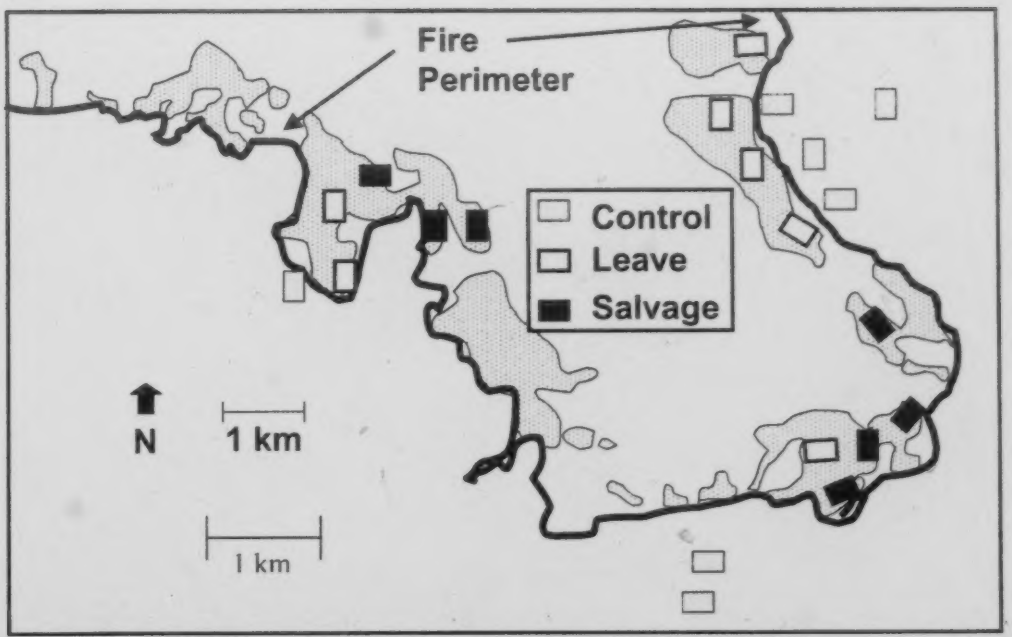

Figure 4.1. Schematic of the Chip Lake Salvage Harvesting Study design. The study area is located in the lower foothills ecoregion of Alberta, north of Cynthia, Alberta. Dotted areas indicate total area that was salvage harvested within the fire. Sites were established in the winter of 1998/99, after a fire burned through the region in the spring of 1998. Control = not burned and non-harvested, Leave = burned and non-harvested, Salvage = burned and salvage harvested.

4.2.2 Secondary cavity nesting bird surveys

Spot mapping was used to estimate numbers of birds at each site (International Bird Census Committee 1970). Bird surveys were conducted in the winter months, from February through mid-April in 1999 (5 rounds), 2000 (8 rounds), 2001 (4 rounds), and 2003 (3 rounds) and were concentrated around mid-day (12:00 P.M. MST). Every attempt was made to determine the exact location and species of each SCN within the site. Observers walked 2 transect lines along the length of each site (100m spacing between each line walked) and recorded every bird seen or heard. Observers were instructed to spend approximately 90 minutes at each site during every survey. In order to standardise effects associated with multiple observers

(Verner and Milne 1990), visits to each site were rotated among observers within each sampling interval.

While efforts were made to conduct SCN surveys at each site for a standardised 90-minute period during each visit, specific survey lengths varied. To correct for this, all counts of SCN's were transformed to hourly estimates of relative abundance. The results for all rounds in a given year were then averaged over the to calculate the mean number of SCN's observed in each site per hour.

4.2.3 Statistical analysis

Standard errors are reported for all results in this chapter, as means of several plots or rounds were determined within each site prior to analysis by treatment. For all statistical tests, alpha was set at 0.10 to reduce the chances of committing a Type II error, as Type II errors can be more costly than Type I errors in applied conservation research (Johnson 1999). SPSS 10.0 was used to carry out statistical tests (SPSS Inc. 1999).

4.2.3.1 Relative abundance of SCN's between treatments

To answer the first question of whether SCN abundance varies between treatments, I used a repeated measures ANOVA. Data were square root transformed to satisfy normality and equality of variance (Zar 1996). Bonferroni post-hoc analysis was employed to test for differences between individual treatments. Although the repeated measures ANOVA was useful in determining if time since fire was affecting SCN numbers, further analysis was required to answer specific questions with regards to the direction and magnitude of the change (see Section 4.2.3.2).

4.2.3.2 Annual and seasonal changes in SCN abundance relative to the control treatment

In order to test if SCN abundance varied between years and seasons in the leave treatment, relative to the control treatment, I used simple contrasts to compare changes in SCN abundance (Schmiegelow et al. 1997). Annual changes in relative abundance in all sites were determined for 1999 to 2000, 2000 to 2001 and 2001 to 2003.

4.3 Results

During the four winters that I surveyed resident birds in the burned forest sites within Chip Lake Salvage Harvesting Study Area, a total of 6 secondary cavity nesting bird species was observed (Table 4.1). The most abundant species observed in all treatments was the black-capped chickadee (Table 4.1). Summaries of individual species relative abundance by treatment are reported in Appendix 4.1.

Table 4.1. Total numbers of secondary cavity nesting species, by treatment, observed in the Chip Lake Salvage Harvesting Study in the winters of 1999, 2000, 2001, and 2003 (1 year to 5 years post-fire). Control = not burned and non-harvested, Leave = burned and non-harvested, Salvage = burned and salvage harvested.

	Number of Observations in:			
Species	Leave	Control	Salvage	(Total)
Boreal chickadee (*Poecile hudsonicus*)	31	48	1	80
Black-capped chickadee (*Poecile atricapillus*)	82	83	8	173
Red-breasted nuthatch (*Sitta canadensis*)	4	12	None	16
White-breasted nuthatch (*Sitta canolinensis*)	None	6	None	6
Brown creeper (*Certhia americana*)	23	36	None	59
Northern hawk owl (*Surnia ulula*)	5	None	2	7
(All Species)	145	185	11	341

4.3.1 Relative abundance of SCN's between treatments

Relative abundance of secondary cavity nesting species was significantly different between treatments and significantly different within treatments between years (Figure 4.2.; ANOVA, treatment: $F = 64.63$, $p < 0.001$; year: $F = 21.01$, $p < 0.001$; treatment*year: $F = 5.32$, $p = 0.004$). In all years, control sites supported the greatest relative abundance of SCN's, followed by leave sites, and salvage harvested sites (Bonferroni; $p<0.01$ in all three post-hoc comparisons). Further exploration of the interaction between year and treatment is reported in Section 4.3.2.

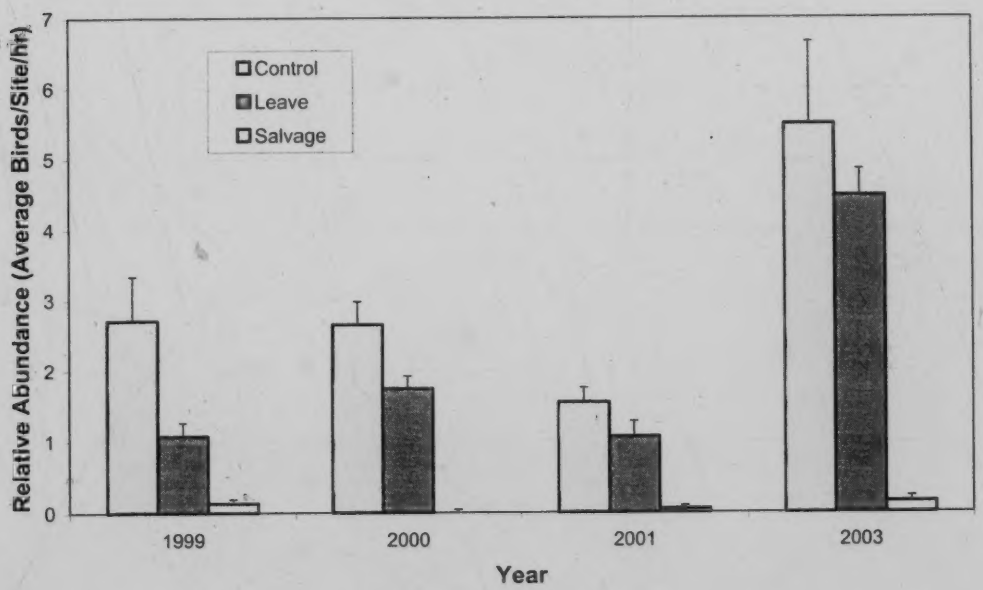

Figure 4.2. Mean relative abundance of secondary cavity nesting species per site by treatment type for 4 winters from 1999 (1st year post-fire) to 2003 (5th year post-fire) in the Chip Lake Salvage Harvesting Study Area. Error bars represent one standard error. Control = not burned and non-harvested, Leave = burned and non-harvested, Salvage = burned and salvage harvested.

4.3.2 Annual and seasonal changes in SCN abundance relative to the control treatment

Abundances of SCN's increased dramatically from 2001 to 2003 (3rd to 5th winter post-fire). This increase was, however, not unique to the leave sites; similar increases were also recorded in the control sites. Relative to the control treatment, no significant changes from one sampling year to the next were observed in the leave treatment (Table 4.2), suggesting that annual changes occurring in the leave treatment were also occurring in the control treatment and were a function of a larger temporal change in the population, not an effect of the treatment. The increase in SCN's within leave sites in the 1st to 2nd winter post-fire (1999-2000) was the nearest to significance of any interval (p=0.14), but the trend was not maintained in subsequent years.

Table 4.2. Change in leave sites, relative to control sites, on annual change in mean secondary cavity nester abundance. Annual change in winter abundance of secondary cavity nesters was measured for 3 periods, 1999-2000 (1st to 2nd year post fire), 2000-2001 (2nd to 3rd year post fire) and 2001-2003 (3rd to 5th year post fire).

Treatment	**1999 to 2000**		**2000 to 2001**		**2001 to 2003**	
(Sampling Interval)	Change[1]	p-value[2]	Change[1]	p-value[2]	Change[1]	p-value
Leave (Winter)	+	0.14	-	0.69	+	0.36

[1] Change in relative abundance relative to that observed in the controls.

4.4 Discussion

Limited information exists on habitat use by winter resident bird species in Canada. Of particular concern is the potential effect that industrial forest harvesting may have on these species. In order to gain a better understanding of how post-fire salvage harvesting may alter habitat quality for winter resident species, I studied secondary cavity nesting winter residents for 4 winters after fire in burned, burned and salvage harvested, and unburned and non-harvested sites. My primary objectives

were 1) to determine if abundances of SCN's were significantly lower in salvage harvested sites and 2) to determine if abundances of SCN's increased for the first several years after fire, in response to an assumed increase in availability of winter roost sites, created by *Picoides* woodpeckers. SCN's were found in significantly lower numbers in the salvage harvested sites when compared to the leave and control sites, with the greatest abundances being recorded in the control (unburned and non-harvested) sites. The absence of SCN's in salvage harvested sites indicates that these sites are of little to no value to this group of resident species in the winter months. Abundances of SCN's did increase dramatically in the 5th winter following fire, but this increase was recorded in both the control and leave sites. Relative to the control sites, no significant changes were recorded in SCN's for any period within the first 5 years after fire. This suggests that, although SCN's are present in recently burned forests, they do not appear to be responding to increases in available roosting sites created by *Picoides* woodpeckers.

4.4.1 Relative abundance of SCN's between treatments

Previous studies have found SCN's in greater abundance in burned and non-harvested forests in the summer months when compared to salvage harvested forests (Caton 1996, Stambaugh 2003). This study indicates that salvage harvested sites also support significantly lower abundances of SCN's in the winter months. Further results suggest that unburned forests contain significantly greater numbers of secondary cavity nesters than burned forests in the winter. Kriesel and Stein (1999) suggested that shrub and branch foraging species (such as black-capped and boreal chickadees) are regulated by food in the winter months and that unburned forests contain greater densities of tall shrubs, resulting in greater food abundance in unburned forests when compared with burned forests. Although I did not test this directly, food regulation may explain differences in bird abundance within my study area, as my prediction that SCN's would increase as a result of *Picoides* woodpecker cavity creation did not hold true (see section 4.4.2 for discussion). Tall shrubs, important for species such as chickadees for foraging, were consistently observed to

have greater percent cover and taller mean height in the control treatments, relative to the leave treatment (Stambaugh 2003). Food regulation of SCN numbers in winter is also consistent with other studies of individual SCN's (i.e. chickadees; Brittingham and Temple 1988, Desrochers et al. 1988). Additional thermal cover, provided by increased foliage from live trees in unburned sites, may also contribute to higher abundances of SCN's compared to burned forests. In summary, unburned forests have the greatest value to secondary cavity nesting species in winter; recently burned forests have moderate value for the same species, and salvage harvested forests have little to no value to SCN's.

4.4.2 Annual and seasonal changes in SCN abundance relative to the control treatment

An increase in regional populations of secondary cavity nesters probably occurred within the larger geographic area of my study site (near Cynthia, Alberta) between 2001 and 2003. Smith (1991) suggests that black capped chickadees (the most abundant SCN in my study area) are irruptive due to changes in weather and food availability. While abundances of SCN's increased in the leave sites from 2001 to 2003, it was not related to treatment, as similar trends were observed in the control sites, and no significant changes in SCN abundances within the leave sites were seen between any years, relative to the control sites. I conclude that presumed increases in the availability of winter roosting sites, created by elevated numbers of *Picoides* woodpeckers in burned sites, do not translate into increased numbers of SCN's in the winter months in burned boreal mixedwood forest.

4.4.3 Study limitations

Abundances of secondary cavity nesters in burned forests were examined for only 5 years post-fire; however, legacies of snags remain for decades post-fire. In order to gain a full understanding of how secondary cavity nesters use burned forests in the winter months, longer-term studies are recommended.

Other possibly factors that may limit secondary cavity nesters in winter months (such as food availability) were not directly examined in my research, and could be addressed with future research.

4.4.4 Relevance to forest management

Salvage harvesting of recently burned forests is common practice in the boreal forests of Canada, and is most frequently conducted in the winter months. My research suggests that forests that have been salvage harvested have little to no value in winter to resident secondary cavity nesting birds, compared to burned and non-harvested forests. However, burned forests maintain significant value for these species, a pattern that has not previously been documented. I suggest that salvage harvesting plans should consider the value of burned forest for secondary cavity nesting birds, as these species are among those considered most at threat by current forest management activities (Schmiegelow and Monkkonen 2002). While older, unburned forests supported greater numbers of SCN's than burned forests during winter, the availability of older forests will be dramatically reduced in this system in coming decades (Schneider et al. 2003). Therefore, maintaining some residual areas of older, burned forest free from salvage harvest is consistent with a precautionary approach (e.g. Dovers and Handmer 1995) to resource management.

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Chapter 5: Thesis conclusions and management implications

5.1 Summary and conclusions

The most prominent and consistent result through every component of this research was the negative impact that salvage harvesting had on *Picoides* woodpeckers and the ecological communities with which they are associated. *Picoides* woodpeckers, potential prey (bark and boring beetles), potential nesting/roosting sites (deciduous snags), and secondary cavity nesters were all found in significantly lower numbers within the salvage harvested forests (salvage) when compared to burned, non-harvested (leave) and unburned, non-harvested forests (control).

Consistent with previous studies (Caton 1996, Hitchcox 1996), I found *Picoides* woodpeckers in greater abundance in the leave sites when compared with both the control and salvage sites, indicating not only a negative impact of salvage harvesting, but reinforcing the overall value of burned forests for woodpecker populations. Adding to previous research, I found that deciduous snag densities and indices of abundance of bark/boring beetles were all important in explaining abundances of three-toed, black-backed, and hairy woodpeckers and deciduous snag densities were important in explaining abundances of downy woodpeckers (Chapter 2).

Burns are highly variable, resulting in non-uniform post-fire habitat (Alexander 1982). My research suggests that *Picoides* woodpeckers and potential species (bark and boring beetles; buprestids, scolytids, and cerambycids) vary in abundance with burn severity. Within my study area, forests of high burn severity appear to be more suitable for hairy and downy woodpeckers in the winter. Three-toed woodpeckers, black-backed woodpeckers, buprestids, scolytids, and cerambycids varied with burn severity, but showed no clear relationship with burn severity, other than the notable result that all these species were found in low levels in sites with low burn severity (Chapter 3). These post-fire specialist species may be

attracted to burn forests for the overall increased availability of deciduous snags and potential prey (Chapter 2), but once within the burned forest, appear not to be locally limited by prey or nest site availability. This supposition is supported by other research that suggests that post fire specialist woodpeckers (notably the black-backed woodpecker) are attracted to post fire forests for the food resource, but are not food limited once in a recently burned forest (Powell 2000).

Within the Chip Lake Study Salvage Harvesting Study Area, recently burned forests supported greater abundances of *Picoides* woodpeckers in the winter months when compared to the breeding season, and abundances declined with time-since fire, relative to the control treatment (Chapter 2). I speculate that secondary cavity nesters would increase in the winter months with time since fire, as a direct result of the increased roost sites created by *Picoides* woodpeckers; however, relative to the control treatment, no significant changes were noted (Chapter 4). Food availability may be relatively more limiting for birds in the winter months (Newton 1998), and I suggest that recently burned forests are of great importance for *Picoides* woodpeckers in the winter months immediately post-fire. While the value for woodpeckers may diminish with time since-fire, these forests remain important for other species of birds, notably secondary cavity nesters. Overall, forests originating from fire are important for a variety of species and salvage harvested sites do not support the same values. This is consistent with other research in the boreal forest, which has shown some species of birds to be present in post-fire forests and absent in harvested forests, for several decades after the disturbance event (Hobson and Schieck 1999).

5.2 Management implications

Through several policy statements, the government of Alberta has demonstrated a commitment to managing the forest resources in an ecological manner. Part of this commitment includes maintenance of the ecological integrity of the forest through a process of adaptive management (Alberta Environmental Protection 2001). Consistent with this philosophy, the Chip Lake Salvage Harvesting Study was established to improve our knowledge of the effects of salvage harvesting on avian species and adapt management practices accordingly. As a result, this

research contributed to the creation of draft guidelines for salvage harvest in Alberta (Alberta Environmental Protection 2002). Although this document addresses many of the issues associated with salvage harvesting, its focus is largely on maintaining timber supply to companies and recommendations are often vague. As a result, there is still a need for refinement of salvage harvesting in Alberta to maintain multiple values. The following recommendations are based on my research in the Chip Lake Fire, some of which have been incorporated in the 2002 Alberta Environmental Protection document. Some remain to be addressed.

5.2.1 Recommendation #1 – Maintain a proportion of burned merchantable timber within each burned landscape.

Our results suggest that salvage harvested sites do not support populations of *Picoides* woodpeckers, their prey (bark and boring beetles), or secondary cavity nesters. When planning for salvage harvesting of a burned area, I recommend that a proportion of merchantable timber be set aside for maintenance of wildlife habitat. It is unknown what the exact size and configuration of non-harvested forest should be (see Section 5.3.1), as this study did not address those questions.

5.2.2 Recommendation #2 – Maintain forest stands of varying burn severities within each burned landscape.

Our results suggest that the importance of burned forest to species of woodpeckers and bark/boring beetles varies with burn intensity. In order to maintain suitable habitat for all species (thus maintaining high levels of biodiversity), salvage harvesting plans within each fire should ensure the maintenance of multiple forest stands with varying levels of burn severity.

The goal of many forest managers over the last decade has been to visually approximate post-fire forests by leaving snags and residual green-tree patches within unburned and harvested areas. However, it is becoming increasingly clear that immediate post-harvest forests are compositionally and structurally different than immediate post-fire forests, particularly in relation to bird communities (e.g., Hobson

and Schieck 1999). The most striking difference is the high densities of snags in burned sites, which are not present in harvested sites. High densities of snags equate to high densities of bark and boring beetles and high abundances of avian species such as *Picoides* woodpeckers (Chapter 2). In the context of forest harvesting, the only way to functionally mimic post-fire forests is not to clear-cut harvest, but rather to notch and girdle trees within a forest, creating a stand of recently killed trees that may be colonized by bark and boring beetles and woodpeckers (Aulen 1991). Considering that girdling and notching trees in a forest stand for the sole purpose of creating wildlife habitat is unrealistic and unnecessary in the North American boreal forest, the best option is to plan salvage harvest so as to maintain portions of burned forest on the landscape.

5.3 Future research

It is important to remember that this study examined woodpecker ecology and response to salvage harvesting within only one fire, and focussed on one forest type (older, deciduous dominated forest). Future studies should examine response by post-fire species in multiple fires and forest types, in order to increase confidence in these results, and to provide a basis for extrapolation over larger areas of the boreal forest.

As with every research project working in an adaptive management framework, my results have led to further questions that require answers in order to guide long-term planning. I recommend that future forest management efforts in post-fire forests aim to salvage harvest in conjunction with research efforts to answer the following questions:

5.3.1 What size and configuration of post-fire forest is required on the landscape to ensure long-term persistence of post-fire species?

Within the Chip Lake Study Salvage Harvesting Study Area, all study areas were of the same shape and the same size (10ha). As a result, I was unable to draw any conclusions with regards to how large an area of recently burned forest is required for the local persistence of *Picoides* woodpecker populations. Future studies could address this question through experimentally salvage harvesting areas, leaving

non-harvested sites of varying shapes and sizes. Determining the requirements for persistence of these species at regional scales is best addressed through spatial simulation modelling.

5.3.2 Does unmerchantable timber have the same value as merchantable timber for post-fire species?

I restricted my research to older deciduous dominated mixedwood (merchantable) forests. As a result, I could not draw any conclusions with regards to the value of non-merchantable forests (i.e. black spruce and younger forests) to *Picoides* woodpeckers and potential prey. This is an important question, because salvage harvesting is presently restricted to merchantable forests in Alberta. If non-merchantable forests have equal value to post-fire species, salvage harvesting on a landscape level may not be as detrimental to woodpecker populations as anticipated.

5.3.3 Is there value in snags retained within salvage-harvested areas?

Within the salvage harvested sites within the Chip Lake Salvage Harvesting Study Area, few to no snags were retained. As a result, I was not able to study the potential value that snag retention within a salvage harvested site may have. Anecdotally, I can report that the limited bird records for salvaged sits occurred in conjunction with the few retained snags. Future research should aim to retain different levels of snags within harvested sites, and assess the value that snags may have to post-fire species if retained in salvage harvested sites.

5.3.4 Is there a spillover effect from burned forests into adjacent unburned forests?

Within my research sites, I found *Picoides* woodpeckers to decline with time-since fire in both the control and leave treatments, although the magnitude of the decline differed. Because of the adjacency of my control sites to the burn perimeter, I could not rule out the possibility that woodpecker numbers were inflated in the control sites as a direct result of proximity to the burn. Future research should address this question by placing control sites at varying distances from burn perimeters.

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APPENDIX 1: Additional woodpecker tables.

Appendix 1.1. Mean numbers of woodpeckers/site (+/- SE) recorded in each treatment by species and sampling interval for 5 years after fire (1999-2003) in the Chip Lake Salvage Harvesting Study Area. P-values are for Kruskal-Wallis Tests for differences between treatments for each species and sampling interval.

Species	Sampling Interval	Control	Leave	Salvage	P-value
	1999 winter	0.24 (0.10)	0.92 (0.15)	0.11 (0.07)	0.015
	1999 breeding season	0.07 (0.03)	0.55 (0.16)	0.05 (0.03)	0.002
	2000 winter	0.15 (0.04)	0.19 (0.06)	0.05 (0.04)	0.107
Hairy	2000 territorial establishment	0.06 (0.04)	0.38 (0.12)	0.02 (0.02)	0.001
Woodpecker	2000 breeding season	0.06 (0.03)	0.33 (0.09)	0.01 (0.00)	0.002
	2001 winter	0.02 (0.02)	0.32 (0.08)	0 (0)	0.001
	2001 territorial establishment	0.13 (0.05)	0.28 (0.12)	0.05 (0.03)	0.246
	2001 breeding season	0.15 (0.04)	0.25 (0.07)	0 (0)	0.002
	2003 winter	0.07 (0.05)	0.04 (0.04)	0 (0)	0.348
	1999 winter	0.04 (0.04)	0.83 (0.25)	0 (0)	0.008
	1999 breeding season	0.01 (0.01)	0.04 (0.02)	0.02 (0.02)	0.477
	2000 winter	0.04 (0.03)	0.40 (0.10)	0 (0)	0.001
Downy	2000 territorial establishment	0.12 (0.07)	0.08 (0.05)	0.02 (0.02)	0.424
Woodpecker	2000 breeding season	0 (0)	0.03 (0.02)	0 (0)	0.122
	2001 winter	0.02 (0.02)	0.02 (0.02)	0.02 (0.02)	1.000
	2001 territorial establishment	0.17 (0.08)	0.04 (0.03)	0 (0)	0.138
	2001 breeding season	0 (0)	0.01 (0.01)	0 (0)	0.368
	2003 winter	0.04 (0.04)	0.04 (0.04)	0 (0)	0.589
	1999 winter	0.23 (0.17)	1.36 (0.34)	0.16 (0.10)	0.013
	1999 breeding season	0.01 (0.01)	0.22 (0.13)	0.04 (0.02)	0.141
	2000 winter	0.10 (0.05)	0.53 (0.19)	0.01 (0.01)	0.007
Three-toed	2000 territorial establishment	0 (0)	0.25 (0.10)	0.02 (0.02)	0.008
Woodpecker	2000 breeding season	0.01 (0.01)	0.11 (0.04)	0.02 (0.02)	0.036
	2001 winter	0.03 (0.03)	0.14 (0.06)	0 (0)	0.042
	2001 territorial establishment	0.04 (0.04)	0.14 (0.07)	0.05 (0.05)	0.430
	2001 breeding season	0.02 (0.01)	0.07 (0.04)	0 (0)	0.138
	2003 winter	0.04 (0.04)	0 (0)	0 (0)	0.368
	1999 winter	0 (0)	0.65 (0.19)	0.09 (0.06)	0.011
	1999 breeding season	0 (0)	0.05 (0.05)	0.02 (0.02)	0.589
	2000 winter	0.01 (0.01)	0.24 (0.05)	0.01 (0.01)	0.002
Black-backed	2000 territorial establishment	0.02 (0.02)	0.05 (0.03)	0.02 (0.02)	0.618
Woodpecker	2000 breeding season	0.01 (0.01)	0.04 (0.03)	0.06 (0.03)	0.150
	2001 winter	0 (0)	0.07 (0.05)	0.03 (0.03)	0.332
	2001 territorial establishment	0 (0)	0.04 (0.03)	0 (0)	0.122
	2001 breeding season	0 (0)	0.02 (0.01)	0.01 (0.01)	0.332
	2003 winter	0.02 (0.02)	0.09 (0.06)	0 (0)	0.296

APPENDIX 2: Additional snag tables.

Appendix 2.1. Mean numbers of snags/ha/site (+/- SE) recorded in each treatment by snag type and year in the Chip Lake Salvage Harvesting Study Area. P-values are for Kruskal-Wallis Tests for differences between treatments for each snag type and year. Only deciduous snags were used in analysis within this thesis, as coniferous snags were highly correlated with bark and boring beetles.

Snag Type	Year	Control	Leave	Salvage	P-value
Coniferous	1999	2.4 (0.8)	119.6 (43.8)	9.5 (4.6)	0.002
	2000	3.0 (2.4)	136.3 (44.6)	10.7 (5.1)	0.005
	2001	4.2 (1.6)	141.7 (45.1)	6.0 (2.4)	0.003
Deciduous	1999	96.4 (6.8)	124.4 (21.1)	5.4 (2.7)	0.001
	2000	94.1 (9.4)	160.7 (22.5)	4.8 (1.9)	0
	2001	95.2 (23.1)	169.6 (23.1)	7.7 (3.3)	0

APPENDIX 3: Additional invertebrate tables.

Appendix 3.1. Example of how indices of beetle abundance were calculated, using cerambycid data collected from coniferous trees and snags within the 15-23 dbh class at all sites in the Chip Lake Salvage Harvesting Study area for 3 winters post-fire (1999-2001). Indices of abundance are calculated by multiplying the average number per tree in each site by the number of trees and snags recorded at vegetation plots in the same site, and converting the value to a per ha estimate. Zero values are indicative of or a combination of three possibilities: no trees or snags (within the 15-23 dbh class) were recorded in the vegetation plots, no trees or snags were available to sample beetles, and/or trees or snags were sampled but contained no beetles.

Year	Plot	Number of trees and snags sampled for cerambycids	Numbers of cerambycids collected at site	Number of trees and snags recorded in vegetation plots	Numbers/0.24 ha (Numbers / # Trees and Snags Sampled X # Trees and Snags in veg plots)	Index of abundance of cerambycids /ha
1999	C1	3	0.00	42	0.00	0.00
1999	C2	1	0.00	1	0.00	0.00
1999	C3	3	0.00	49	0.00	0.00
1999	C4	3	0.00	21	0.00	0.00
1999	C5	1	0.00	1	0.00	0.00
1999	C6	2	0.00	2	0.00	0.00
1999	C7	3	0.00	52	0.00	0.00
1999	L1	3	22.00	1	7.33	30.56
1999	L2	2	0.00	6	0.00	0.00
1999	L3	3	0.00	24	0.00	0.00
1999	L4	3	0.00	25	0.00	0.00
1999	L5	3	5.00	33	55.00	229.17
1999	L6	3	5.50	31	56.83	236.81
1999	L7	3	0.00	30	0.00	0.00
1999	S1	0	0.00	0	0.00	0.00
1999	S2	0	0.00	2	0.00	0.00
1999	S3	1	0.00	0	0.00	0.00
1999	S4	0	0.00	0	0.00	0.00
1999	S5	0	0.00	0	0.00	0.00
1999	S6	1	0.00	0	0.00	0.00
1999	S7	0	0.00	2	0.00	0.00
2000	C1	3	0.00	37	0.00	0.00
2000	C2	2	0.00	0	0.00	0.00
2000	C3	2	0.00	46	0.00	0.00
2000	C4	3	0.00	29	0.00	0.00
2000	C5	2	0.00	1	0.00	0.00
2000	C6	2	0.00	2	0.00	0.00
2000	C7	3	0.00	52	0.00	0.00
2000	L1	1	0.00	0	0.00	0.00
2000	L2	3	2.50	10	8.33	34.72
2000	L3	3	0.00	28	0.00	0.00
2000	L4	3	1.00	28	9.33	38.89

Appendix 3.1. (cont). Example of how indices of beetle abundance were calculated, using cerambycid data collected from coniferous trees and snags within the 15-23 dbh class at all sites in the Chip Lake Salvage Harvesting Study area for 3 winters post-fire (1999-2001). Indices of abundance are calculated by multiplying the average number per tree in each site by the number of trees and snags recorded at vegetation plots in the same site, and converting the value to a per ha estimate. Zero values are indicative of or a combination of three possibilities: no trees or snags (within the 15-23 dbh class) were recorded in the vegetation plots, no trees or snags were available to sample beetles, and/or trees or snags were sampled but contained no beetles.

Year	Plot	Number of trees and snags sampled for cerambycids	Numbers of cerambycids collected at site	Number of trees and snags recorded in vegetation plots	Numbers/0.24 ha (Numbers / # Trees and Snags Sampled X # Trees and Snags in veg plots)	Index of abundance of cerambycids /ha
2000	L5	3	2.00	36	24.00	100.00
2000	L6	3	2.00	48	32.00	133.33
2000	L7	3	1.00	27	9.00	37.50
2000	S1	0	0.00	0	0.00	0.00
2000	S2	0	0.00	1	0.00	0.00
2000	S3	0	0.00	0	0.00	0.00
2000	S4	0	0.00	0	0.00	0.00
2000	S5	1	0.00	2	0.00	0.00
2000	S6	0	0.00	1	0.00	0.00
2000	S7	1	0.00	2	0.00	0.00
2001	C1	3	0.00	43	0.00	0.00
2001	C2	2	0.00	1	0.00	0.00
2001	C3	3	0.00	64	0.00	0.00
2001	C4	3	0.00	27	0.00	0.00
2001	C5	3	0.00	4	0.00	0.00
2001	C6	2	0.00	4	0.00	0.00
2001	C7	3	0.00	49	0.00	0.00
2001	L1	1	0.00	0	0.00	0.00
2001	L2	3	2.00	7	4.67	19.44
2001	L3	3	0.00	32	0.00	0.00
2001	L4	3	0.00	23	0.00	0.00
2001	L5	3	0.00	30	0.00	0.00
2001	L6	3	9.50	38	120.33	501.39
2001	L7	3	6.50	28	60.67	252.78
2001	S1	0	0.00	0	0.00	0.00
2001	S2	0	0.00	2	0.00	0.00
2001	S3	0	0.00	0	0.00	0.00
2001	S4	0	0.00	0	0.00	0.00
2001	S5	1	0.00	0	0.00	0.00
2001	S6	0	0.00	0	0.00	0.00
2001	S7	0	0.00	1	0.00	0.00

Appendix 3.2. Example of how indices of beetle abundance were calculated, using cerambycid data collected from deciduous trees and snags within the 15-23 dbh class at all sites in the Chip Lake Salvage Harvesting Study area for 3 winters post-fire (1999-2001). Indices of abundance are calculated by multiplying the average number per tree in each site by the number of trees and snags recorded at vegetation plots in the same site, and converting the value to a per ha estimate. Zero values are indicative of or a combination of three possibilities: no trees or snags (within the 15-23 dbh class) were recorded in the vegetation plots, no trees or snags were available to sample beetles, and/or trees or snags were sampled but contained no beetles.

Year	Plot	Number of trees and snags sampled for cerambycids	Numbers of cerambycids collected at site	Number of trees and snags recorded in vegetation plots	Numbers/0.24 ha (Numbers / # Trees and Snags Sampled X # Trees and Snags in veg plots)	Index of abundance of cerambycids /ha
1999	C1	3	1.00	33	11.00	45.83
1999	C2	3	0.00	10	0.00	0.00
1999	C3	3	0.00	30	0.00	0.00
1999	C4	3	0.00	24	0.00	0.00
1999	C5	3	0.00	48	0.00	0.00
1999	C6	3	0.00	40	0.00	0.00
1999	C7	3	0.00	29	0.00	0.00
1999	L1	3	10.00	17	56.67	236.11
1999	L2	2	0.00	15	0.00	0.00
1999	L3	3	0.00	11	0.00	0.00
1999	L4	1	0.00	6	0.00	0.00
1999	L5	3	0.00	6	0.00	0.00
1999	L6	3	1.00	30	10.00	41.67
1999	L7	3	0.00	21	0.00	0.00
1999	S1	0	0.00	0	0.00	0.00
1999	S2	1	0.00	0	0.00	0.00
1999	S3	0	0.00	0	0.00	0.00
1999	S4	0	0.00	0	0.00	0.00
1999	S5	1	0.00	0	0.00	0.00
1999	S6	0	0.00	0	0.00	0.00
1999	S7	0	0.00	0	0.00	0.00
2000	C1	3	0.00	27	0.00	0.00
2000	C2	3	0.00	5	0.00	0.00
2000	C3	3	0.00	22	0.00	0.00
2000	C4	3	0.00	21	0.00	0.00
2000	C5	3	0.50	51	8.50	35.42
2000	C6	3	0.00	37	0.00	0.00
2000	C7	3	0.00	29	0.00	0.00
2000	L1	3	0.00	20	0.00	0.00
2000	L2	3	0.00	13	0.00	0.00
2000	L3	2	0.00	13	0.00	0.00
2000	L4	3	0.00	4	0.00	0.00
2000	L5	3	1.00	9	3.00	12.50
2000	L6	3	0.00	28	0.00	0.00

Appendix 3.2 (cont.). Example of how indices of beetle abundance were calculated, using cerambycid data collected from deciduous trees and snags within the 15-23 dbh class at all sites in the Chip Lake Salvage Harvesting Study area for 3 winters post-fire (1999-2001). Indices of abundance are calculated by multiplying the average number per tree in each site by the number of trees and snags recorded at vegetation plots in the same site, and converting the value to a per ha estimate. Zero values are indicative of or a combination of three possibilities: no trees or snags (within the 15-23 dbh class) were recorded in the vegetation plots, no trees or snags were available to sample beetles, and/or trees or snags were sampled but contained no beetles.

Year	Plot	Number of trees and snags sampled for cerambycids	Numbers of cerambycids collected at site	Number of trees and snags recorded in vegetation plots	Numbers/0.24 ha (Numbers / # Trees and Snags Sampled X # Trees and Snags in veg plots)	Index of abundance of cerambycids /ha
2000	L7	2	0.00	22	0.00	0.00
2000	S1	0	0.00	0	0.00	0.00
2000	S2	0	0.00	0	0.00	0.00
2000	S3	0	0.00	0	0.00	0.00
2000	S4	0	0.00	0	0.00	0.00
2000	S5	0	0.00	0	0.00	0.00
2000	S6	0	0.00	0	0.00	0.00
2000	S7	0	0.00	0	0.00	0.00
2001	C1	3	0.00	34	0.00	0.00
2001	C2	3	0.00	8	0.00	0.00
2001	C3	3	0.00	35	0.00	0.00
2001	C4	3	0.00	20	0.00	0.00
2001	C5	3	5.00	52	86.67	361.11
2001	C6	3	0.00	41	0.00	0.00
2001	C7	3	0.00	21	0.00	0.00
2001	L1	3	0.00	19	0.00	0.00
2001	L2	3	0.00	14	0.00	0.00
2001	L3	3	0.00	18	0.00	0.00
2001	L4	3	0.00	7	0.00	0.00
2001	L5	3	0.00	7	0.00	0.00
2001	L6	3	0.00	26	0.00	0.00
2001	L7	3	0.00	24	0.00	0.00
2001	S1	1	0.00	0	0.00	0.00
2001	S2	0	0.00	0	0.00	0.00
2001	S3	0	0.00	0	0.00	0.00
2001	S4	0	0.00	0	0.00	0.00
2001	S5	0	0.00	0	0.00	0.00
2001	S6	0	0.00	1	0.00	0.00
2001	S7	0	0.00	0	0.00	0.00

Appendix 3.3. Mean biomass (+/- SE) of wood-inhabiting invertebrates, measured as the averaged weight (grams) of invertebrates collected per site in the Chip Lake Salvage Harvesting Study Area. Biomass reflects raw values collected, prior to compensations for differing levels of snag and tree densities in each site. P-values are for Kruskal-Wallis Tests for differences between treatments for each area in the wood where invertebrates were collected and year.

Where Collected	Year	Control	Leave	Salvage	P-value
coniferous surface	1999	0.0012 (0.0012)	0.12 (0.11)	0.0008 (0.0006)	0.056
	2000	0.0015 (0.0015)	0.0708 (0.0242)	0.0215 (0.0143)	0.006
	2001	0 (0)	0.0075 (0.0055)	0.0004 (0.0004)	0.101
deciduous surface	1999	0.0008 (0.0006)	0.0291 (0.0143)	0.0003 (0.0003)	0.006
	2000	0.0014 (0.0009)	0.0348 (0.0196)	0 (0)	0.001
	2001	0.0095 (0.0043)	0.0291 (0.013)	0 (0)	0.002
coniferous wood	1999	0 (0)	0.41 (0.3)	0.0016 (0.0016)	0.008
	2000	0 (0)	0.67 (0.34)	0.003 (0.003)	0.008
	2001	0 (0)	0.27 (0.14)	0 (0)	0.01
deciduous wood	1999	0.0526 (0.0471)	0.0024 (0.0022)	0.0017 (0.0017)	0.581
	2000	0.0071 (0.0071)	0.005 (0.0022)	0 (0)	0.02
	2001	0.0983 (0.0983)	0.0049 (0.0024)	0 (0)	0.02
total (sum of all above values)*	1999	0.0546 (0.0476)	0.5615 (0.4137)	0.0044 (0.0026)	0.069
	2000	0.01 (0.0078)	0.7849 (0.3554)	0.0245 (0.0168)	0.002
	2001	0.1077 (0.0987)	0.3159 (0.1398)	0.0004 (0.0004)	0.002

Appendix 3.4. Mean number of beetles (adult and larvae) per site (+/- SE) in the Order Coleoptera, collected in different wood locations in the Chip Lake Salvage Harvesting Study Area. Numbers reflect raw values collected, prior to compensations for differing levels of snag and tree densities in each site. P-values are for Kruskal-Wallis Tests for differences between treatments for each area in the wood where invertebrates were collected and year.

Where Collected	Year	Control	Leave	Salvage	P-value
coniferous surface	1999	0.4 (0.3)	70.6 (60.1)	0.4 (0.3)	0.037
	2000	0.3 (0.3)	165.4 (121.0)	2.3 (1.5)	0.002
	2001	0 (0)	3.2 (1.8)	0.4 (0.4)	0.101
deciduous surface	1999	2 (1.6)	8.4 (3.4)	0.1 (0.1)	0.012
	2000	0.7 (0.5)	14.3 (6.6)	0 (0)	0.003
	2001	0.7 (0.6)	5.8 (2.6)	0 (0)	0.004
coniferous wood	1999	0 (0)	7.6 (4.1)	0.1 (0.1)	0.027
	2000	0 (0)	3.1 (1.7)	0.1 (0.1)	0.101
	2001	0 (0)	2.1 (1.6)	0 (0)	0.037
deciduous wood	1999	0.4 (0.3)	1.7 (1.4)	0.1 (0.1)	0.474
	2000	2.4 (2.4)	3.9 (1.9)	0 (0)	0.02
	2001	0.7 (0.7)	3.4 (1.2)	0 (0)	0.014

Appendix 3.5. Mean number of non-Coleopteran invertebrates per site (+/- SE), collected in different wood locations in the Chip Lake Salvage Harvesting Study Area. Numbers reflect raw values collected, prior to compensations for differing levels of snag and tree densities in each site. P-values are for Kruskal-Wallis Tests for differences between treatments for each area in the wood where invertebrates were collected and year.

Where Collected	Year	Control	Leave	Salvage	P-value
coniferous surface	1999	0 (0)	54.7 (47.7)	0 (0)	0.002
	2000	0 (0)	36.4 (20.8)	19.4 (19.4)	0.004
	2001	0 (0)	0.4 (0.4)	0 (0)	0.368
deciduous surface	1999	0 (0)	10.4 (3.5)	0 (0)	0.002
	2000	1 (0.4)	32.1 (17.4)	0 (0)	0.012
	2001	30.7 (23.3)	34.6 (20.7)	0 (0)	0.057
coniferous wood	1999	0 (0)	3.6 (2.3)	0 (0)	0.01
	2000	0 (0)	63.3 (61.1)	0 (0)	0.01
	2001	0 (0)	3.3 (1.9)	0 (0)	0.037
deciduous wood	1999	0.8 (0.6)	0 (0)	1.6 (1.6)	0.365
	2000	1.3 (1.3)	0.4 (0.3)	0 (0)	0.365
	2001	0.1 (0.1)	0.3 (0.2)	0 (0)	0.329

Appendix 3.6. Mean number wood-penetrating galleries created by bark/boring beetles per site (+/- SE) observed in the log samples collected in the Chip Lake Salvage Harvesting Study Area. Numbers reflect raw values collected, prior to compensations for differing levels of snag and tree densities in each site. P-values are for Kruskal-Wallis Tests for differences between treatments for each wood type and year.

Where Collected	Year	Control	Leave	Salvage	P-value
coniferous wood	1999	0.4 (0.3)	50.3 (30.7)	3.7 (3.4)	0.055
	2000	0 (0)	49.3 (21.9)	3.9 (3.9)	0.002
	2001	0 (0)	68.3 (37.9)	0 (0)	0.002
deciduous wood	1999	14.9 (11.6)	3.1 (1.8)	0 (0)	0.21
	2000	3.6 (2.8)	10.7 (5.4)	0 (0)	0.025
	2001	5.7 (5.7)	20.6 (4.2)	0 (0)	0.001

Appendix 3.7. Mean number of snags, averaged for six 11.3 m radius vegetation plots at each site, showing insect attack per site (+/- SE) in the Chip Lake Salvage Harvesting Study Area. P-values are for Kruskal-Wallis Tests for differences between treatments for each tree type and year.

Tree type	Year	Control	Leave	Salvage	P-value
	1999	0.43 (0.2)	22.29 (8.04)	1 (0.53)	0.002
coniferous	2000	0.71 (0.57)	25 (8.76)	1.43 (0.95)	0.003
	2001	0.71 (0.36)	32.71 (10.81)	1.14 (0.59)	0.005
	1999	11.86 (1.22)	9.86 (2.28)	0.43 (0.3)	0.001
deciduous	2000	14.43 (2.15)	16.29 (2.46)	0.29 (0.18)	0.001
	2001	20.14 (2.34)	35.14 (5)	1.57 (0.87)	0

Appendix 3.8. Mean number of invertebrates collected per site in wood samples (+/- SE), grouped by lowest identifiable taxa and year collected in the Chip Lake Salvage Harvesting Study Area. Numbers reflect raw values collected, prior to compensations for differing levels of snag and tree densities in each site.

Taxonomic grouping of	Year	Control	Leave	Salvage
CERAMBYCID LARVAE	1999	0.14 (0.378)	4.5 (7.72)	0
	2000	0.07 (0.19)	1.36 (1.18)	0
	2001	0.71 (1.89)	2.36 (3.45)	0
CERAMBYCID ADULT	1999	0	1.71 (4.11)	0
	2000	0	0	0
	2001	0	0.21 (0.57)	0
BUPRESTID LARVAE	1999	0.29 (0.76)	15.93 (16.04)	0.57 (1.13)
	2000	0	8.71 (9.79)	1.29 (1.22)
	2001	0	1.86 (2.37)	0.29 (0.76)
BUPRESTID ADULT	1999	0	0	0
	2000	0	0	0
	2001	0	0	0
SCOLYTID LARVAE	1999	0	3.71 (7.52)	0
	2000	0	128.43 (315.57)	0
	2001	0	0.43 (1.13)	0.14 (0.38)
SCOLYTID ADULTS	1999	0.29 (0.76)	56.07 (132.56)	0.14 (0.38)
	2000	0.43 (0.79)	25.21 (47.22)	0
	2001	0	1.57 (3.31)	0
CUCUJIDAE LARVAE	1999	0	1.36 (1.75)	0
	2000	0	7.43 (12.78)	0.14 (0.38)
	2001	0	3.5 (4.08)	0
CUCUJIDAE ADULTS	1999	0	0	0
	2000	0	0.14 (0.38)	0
	2001	0	0.14 (0.38)	0
OTHER COLEOPTERA LARVAE	1999	0.14 (0.38)	2.86 (4.63)	0.14 (0.38)
	2000	2.64 (6.56)	10.57 (15.90)	0.43 (0.79)
	2001	0.57 (1.51)	2.14 (3.53)	0
OTHER COLEOPTERA ADULTS	1999	2 (4.14)	2.14 (3.237)	0
	2000	0.29 (0.76)	4.86 (3.42)	0.57 (1.51)
	2001	0.14 (0.39)	2.21 (1.93)	0

Appendix 3.8. (Cont.) Mean number of invertebrates collected per site in wood samples (+/- SE), grouped by lowest identifiable taxa and year collected in the Chip Lake Salvage Harvesting Study Area. Numbers reflect raw values collected, prior to compensations for differing levels of snag and tree densities in each site.

Taxonomic grouping of	Year	Control	Leave	Salvage
HEMITERA	1999	0	1.43 (2.51)	0
	2000	0.14 (0.38)	1.29 (0.95)	0
	2001	0	0.29 (0.76)	0
HYMENOPTERA	1999	0	1.57 (2.44)	0
	2000	0.14 (0.38)	64.36 (158.27)	0
	2001	0	2.21 (3.49)	0
ACARIANA	1999	0	0.14 (0.38)	0
	2000	0	0	0
	2001	0	0	0
ARANEIDA	1999	0	0.14 (0.38)	0
	2000	0	0.71 (1.25)	0
	2001	0	0	0
DIPTERA	1999	0	34.71 (63.52)	0
	2000	0.29 (0.49)	52.14 (83.58)	18.07 (47.81)
	2001	30.57 (61.66)	33.86 (54.12)	0
LEPIDOPTERA	1999	0	0	0
	2000	0	0.29 (0.76)	0
	2001	0	0	0
CHILOPODA	1999	0	0	0
	2000	0.14 (0.38)	0	0
	2001	0.14 (0.38)	0	0
GASTROPODA	1999	0	0	0
	2000	1 (2.64)	0.14 (0.38)	0
	2001	0	0.14 (0.38)	0
THYSANOPTERA	1999	0	0	0
	2000	0	1.86 (3.29)	0
	2001	0	0	0
NEUROPTERA	1999	0	0	0
	2000	0	0.14 (0.38)	0
	2001	0	0	0

Appendix 3.8. (Cont.) Mean number of invertebrates collected per site in wood samples (+/- SE), grouped by lowest identifiable taxa and year collected in the Chip Lake Salvage Harvesting Study Area. Numbers reflect raw values collected, prior to compensations for differing levels of snag and tree densities in each site.

Taxonomic grouping of	Year	Control	Leave	Salvage
UNKNOWN	1999	0.79 (1.68)	30.71 (65.14)	1.57 (4.16)
	2000	0.57 (0.79)	11.29 (9.74)	1.29 (3.40)
	2001	0.14 (0.38)	2.04 (2.76)	0

APPENDIX 4: Additional weak and secondary cavity nesting species tables.

Appendix 4.1. Mean numbers of secondary cavity nesting species/site (+/- SE) recorded in each treatment by species for 4 winter sampling intervals. P-values are for Kruskal-Wallis Tests for differences between treatments for each species and sampling interval.

Species	Winter Sampling Interval	Control	Leave	Salvage	P-value
	1999	1.53 (0.48)	0.29 (0.09)	0 (0)	0.003
Boreal	2000	0.49 (0.22)	0.30 (0.09)	0 (0)	0.030
Chickadee	2001	0.17 (0.06)	0.10 (0.06)	0 (0)	0.095
	2003	1.17 (0.42)	0.68 (0.13)	0.04 (0.04)	0.014
	1999	0.77 (0.30)	0.71 (0.10)	0.13 (0.06)	0.015
Black-capped	2000	1.77 (0.38)	1.18 (0.21)	0.07 (0.04)	0.001
Chickadee	2001	1.06 (0.13)	0.82 (0.15)	0.03 (0.03)	0.001
	2003	3.52 (0.69)	3.52 (0.35)	0.08 (0.05)	0.001
	1999	0.14 (0.10)	0 (0)	0 (0)	0.041
Red-breasted	2000	0.03 (0.03)	0 (0)	0 (0)	0.368
Nuthatch	2001	0.03 (0.03)	0 (0)	0 (0)	0.368
	2003	0.53 (0.29)	0.21 (0.14)	0 (0)	0.168
	1999	0 (0)	0 (0)	0 (0)	1.000
White-breasted	2000	0.10 (0.05)	0 (0)	0 (0)	0.037
Nuthatch	2001	0 (0)	0 (0)	0 (0)	1.000
	2003	0.04 (0.04)	0 (0)	0 (0)	0.368
	1999	0.27 (0.10)	0.08 (0.05)	0 (0)	0.058
Brown	2000	0.26 (0.05)	0.24 (0.06)	0 (0)	0.002
Creeper	2001	0.29 (0.12)	0.07 (0.03)	0 (0)	0.019
	2003	0.19 (0.13)	0.07 (0.04)	0 (0)	0.168
	1999	0 (0)	0 (0)	0 (0)	1.000
Northern	2000	0 (0)	0.03 (0.03)	0 (0)	0.368
Hawk-owl	2001	0 (0)	0.07 (0.05)	0.03 (0.03)	0.296
	2003	0 (0)	0 (0)	0 (0)	0.368